

## 6.0 **LOADING, FATE, AND TRANSPORT FOR SELECT CONTAMINANTS**

**Commented [KK1]:** Due to text modifications, Section 6 Tables, Maps and Figures need to be renumbered.

This section presents an assessment of contaminant loading mechanisms to the study area from external sources as well as in-river processes affecting the concentration, transport, and fate of select contaminants within the study area. Section 6.1 assesses contaminant inputs (external loading) to the study area. Section 6.2 describes fate and transport processes that act on contaminants in abiotic and biotic media within the study area. The discussion of fate and transport processes is grouped by sediment and pore water, surface water, and biotic processes. Section 6.3 provides an evaluation of sedimentation in the upper study area utilizing three sediment core profiles in the navigation channel; two in borrow pits, which are natural sediment traps, and one in a shoal area.

Contaminants evaluated in this assessment are presented in Table 6.0-1. There are separate lists for surface water, stormwater, upland groundwater plumes, atmospheric deposition, and equilibrium partitioning (advective loading from subsurface sediment to surface sediment and from surface sediment to surface water). These lists reflect data availability by media and relevance of the contaminant to the loading mechanism. For example, equilibrium partitioning primarily focuses on hydrophobic contaminants and metals, stormwater and atmospheric deposition contaminants reflect the limited available data sets, and upland plume loading contaminants reflect individual upland plumes.

### 6.1 **EXTERNAL CONTAMINANT LOADING**

Loading is a quantity of mass that passes a boundary over a given time frame. The boundaries for determining external loads include the upstream and downstream river mile designations (RM 1.9 and 11.8), the surface of the river, the river bank, and the surface sediment/subsurface sediment boundary at 30 cm bml. Contaminant masses passing through these boundaries are external loads.

These loading mechanisms represent the combined estimated load from all study area sources for the corresponding pathway. A simplified conceptualization of the external loading pathways (loading terms) and internal transport processes is presented on Figure 6.1-1. Numerical loading estimates were generated for the following external contaminant loads:

- Upstream loading via surface water, including suspended sediment load
- Stormwater runoff
- Permitted non-stormwater point source discharges
- Upland groundwater plume transport to the river

- Atmospheric deposition to the river surface

Loading estimates for upland riverbank erosion, sediment bedload, and overwater releases were not quantified. Contaminant releases from overwater activities, such as sandblasting, painting, material transfer, maintenance, repair, and operations at riverside docks, wharfs, or piers; discharges from vessels; fuel releases; and spills are not considered quantifiable and are not addressed in this section. Releases of this nature are expected to have been more significant historically, prior to improved BMPs. While improved BMPs are likely to have reduced the occurrence of overwater releases significantly, although current and future releases could occur. Due to insufficient available information, no attempt is made in this report to predict and quantify such releases as a current loading term.

Quantification of sediment bedload (rolling, sliding, and saltating of sediment grains) into the study area and associated contaminant transport is not quantified in the HST model because it was assumed that bedload represents a relatively small fraction of the total sediment load entering the study area at RM 11.8. This is because of the lower Willamette River's morphology and the fact that its flows are regulated by upstream control structures. As noted in Section 3, the study area occupies the lower portion of the lower Willamette River where the river widens and has been deepened by dredging. The reach upstream of the study area, from Willamette Falls through downtown Portland, is generally narrower with high velocities, so suspended loads tend to be transported into the study area before settling out. In addition, dams at Willamette Falls (RM 26) and further upstream trap bedload moving downstream from the middle Willamette River to the lower Willamette River. The only significant tributary to the lower Willamette River below RM 26 is the Clackamas River at RM 24.7 and it is a gravel-bed stream.

Due to insufficient chemistry data at multiple shoreline sites, it is not possible at this time to estimate riverbank loading to the river. Further, it is not possible to estimate typical erosion rates or a range of rates that might apply to riverbanks in the study area given the wide range of conditions present. Since contaminant loading from bank erosion is an area-specific condition dependent on both the erodibility and contaminant concentrations at any given bank area, the contaminant loading is more important on a more localized scale rather than as a load to the study area as a whole.

A range of estimates (central estimate and upper- and lower-bounds) is provided for all loading terms for which numerical estimates were generated to give perspective on the uncertainty associated with a given pathway for each contaminant. The estimation approach for each term varies in nature and approach depending on the degree to which loading associated with a given transport pathway could be evaluated using available information. Most, if not all, of the attributes utilized in the loading estimations are based on site-specific measurements and monitoring results. If attributes were either not measurable or site-specific data are not available, literature data or empirical data collected outside of the study area are used. Some assumptions and modeling or calculation techniques may be used in these assessments.

The assessments of external loading terms are intended to illustrate the estimated magnitude and variability in contaminant loads to the study area under typical conditions in an average water year. Assessment of year-to-year temporal variability was not the intent of this analysis. Because every water year is slightly different from the theoretical “average water year,” the analyses include data collected during a range of environmental conditions. This variability is taken into account in the analyses to the extent possible. The approach to assess each term is discussed in Appendix E.

The target scale of assessment of current loading rates is mass per year; however, in many cases, the data set supports calculation of loading estimates at smaller temporal and/or spatial resolution. Where possible and relevant to understanding the system for the purposes of the RI, these more refined loading estimates were generated and are presented and discussed.

#### **6.1.1 Upstream Contaminant Load**

Upstream contaminant load is defined as the mass transport over time of a given contaminant across the upriver study area boundary at RM 11.8. Upstream contaminant loading is subdivided into dissolved and suspended solids fractions. Surface water loading is assessed in this section for typical flow conditions, as well as extreme observed and modeled high-flow conditions.

While upstream loading terms are presented here as dissolved surface water and suspended particulate loads, these loads represent the combined input to the study area from a variety of loading processes in the upstream watershed. These inputs include upstream point sources, upstream stormwater runoff, upstream CSOs, upstream atmospheric deposition, and upstream in-river sources. Distinguishing these individual contributions to the combined upstream load is beyond the scope of this document.

Concentration and flow rate data from the site were used to generate a range of estimates of annual upstream loading rates. Estimates for dissolved, particulate, and total loading rates are presented. Upstream surface water loading at RM 11.8 was estimated based on analytical data collected from sampling transects at RM 16 and 11.

##### **6.1.1.1 Data Sets and Approach**

Upriver surface water loading rates were estimated based on Round 2A and 3A surface water contaminant concentration data from the RM 16 transect sampling location and USGS flow information from RM 12.8 (Morrison Bridge Station 14211720). The annual flow regimes (Section 3) for the site have two distinct periods of flow: a high-flow and a low-flow condition. The approach discussed here, and described in more detail in Appendix E2, describes how the data was apportioned to represent a single external load.

To differentiate loads associated with high-flow and low-flow conditions during a typical flow year, the fraction of a typical water year that is described by each flow regime was determined. Since the USGS gage station and flow conditions during data

collection compared well, the 28-year hydrograph was considered adequately representative for use as the basis for defining the high-flow:low-flow volume ratio of 1.07 for a typical year.

The surface water analytical data set was apportioned to estimate representative concentration ranges for high-flow and low-flow conditions. Three surface water sampling events from the Round 2A sampling effort and four surface water sampling events from the Round 3A sampling effort provided the analytical data for the surface water loading calculations. Of these seven sampling events, four occurred during low-flow conditions (<50,000 cfs), two were during high-flow conditions (>50,000 cfs), and one was during a low-flow stormwater event (active runoff to the study area with river flow rate <50,000 cfs). Individual data points associated with the RM 16 transect were averaged to represent a single concentration per transect per sampling event (see Appendix E, Section 2.2.2 for details), and minimum, mean, and maximum concentrations at each transect was calculated for both high-flow and low-flow conditions.

Upriver loading rates were estimated as the product of the contaminant concentrations and the flow volumes associated with the high-flow and low-flow portions of the hydrograph. Lower, central, and upper estimates of high-flow loading were estimated for each transect by multiplying the minimum, mean, and maximum concentrations, respectively, by the total annual flow volume estimated for high-flow conditions. Low-flow rate loading estimates for each transect was estimated using the minimum, mean, and maximum of averaged concentrations and the estimated flow volume for low-flow conditions. The range of annual mass loading rate estimates were generated by summing the fractional loading contributions estimated for high-flow and low-flow conditions at the given transect.

There are no surface water sample results available from RM 11.8, which represents the upstream boundary of the study area. Therefore, high-flow and low-flow concentrations at RM 11.8 were estimated by combining data from RM 16 with selected data from RM 11. Because the surface water samples collected on the east side of RM 11 appear to have been influenced by one or more source areas of contaminants, these results are considered not representative of water quality entering the Site at the upstream boundary. Prior to combining the data, the RM 11 data set was assessed for each contaminant to determine whether the data represented the same population of upstream data as that sampled at RM 16. This approach assumes that the surface water concentrations at RM 11.8 would be similar to those at RM 16 than those at RM 11, recognizing that although there are additional sources between RM 11.8 and 16, the proximity of likely sources to the RM 11 transect are expected to have the larger effect. A graphical and statistical comparison of the contaminant data from RM 16 and 11 was conducted for each selected contaminant and is described in Appendix E, Section 2.2.2.1.

#### 6.1.1.2 Uncertainty Associated with Surface Water Loading Estimates

Uncertainty associated with the surface water loading estimates is related primarily to the adequacy and representativeness of the analytical data set. The data sets are derived from grab samples, not time-weighted composites. Further, a limited number of samples were collected under a limited number of flow conditions. This prohibits a thorough understanding of temporal and flow variability in surface water quality and is an important source of uncertainty. The magnitude and direction of bias on loading estimates is unknown.

#### 6.1.1.3 Annual Upstream Loading

Table 6.1-2 presents the range of total (dissolved plus particulate) annual upstream loading estimates for each contaminant evaluated. Figure 6.1-2 presents total upriver surface water loading estimates for total PCB congeners and selected individual PCB congeners. The total PCBs loading estimates show higher aggregate loads during the low-flow period of the year as compared to the high-flow period. Total PCBs show significant contributions of particulate-associated concentrations to the total surface water PCB loads for most flow conditions (Figure 6.1-3). These patterns in flow conditions and particulate/dissolved ratios are also generally apparent in the individual congener data sets.

Upriver loads for total PCDD/Fs and TCDD TEQ (Figures 6.1-4 and 6.1-5) are primarily associated with particulate matter. The relative contributions to the annual load from high-flow and low-flow periods are comparable (Figure 6.1-4).

The upriver loads for DDX compounds (Figures 6.1-6 and 6.1-7) indicate the loads are generally higher in the particulate fraction, as typically seen for the other hydrophobic contaminants, but the patterns with surface water flow regime differ. The annual aggregate upstream load of DDX compounds associated with the high-flow period is consistently higher than that associated with the low-flow period of the year. Further, the 4,4'-isomers of the DDX components compose the majority of the DDX upstream load, with DDT being the greatest fraction, and DDD being the smallest fraction of the DDX.

Upstream surface water loads of total PAHs are greater (approximately an order of magnitude) than total cPAHs (Figure 6.1-8). LPAHs generally exhibit greater solubility than HPAHs; cPAHs are primarily classified as HPAH compounds. The annual aggregate load of LPAHs and HPAHs associated with the high-flow period is higher than that associated with the low-flow period of the year. On a daily basis, total PAHs loads are higher during the high-flow period than during the low-flow period. LPAHs show higher fractions of dissolved as compared to particulate load (Figure 6.1-9). In contrast, HPAHs are generally more hydrophobic, and show higher fractions in the particulate load as compared to dissolved load.

Upstream loading rate ranges for BEHP and hexachlorobenzene are presented on Figures 6.1-10 and 6.1-11. The total annual BEHP load is almost exclusively associated

with high-flow periods. The high-flow contribution for hexachlorobenzene is also higher than the low-flow contribution. Hexachlorobenzene shows consistent fractions of particulate and dissolved contributions to the total load under all flow conditions, with the particulate fraction making up roughly 15 to 20 percent of the total load.

Upstream surface water loading rate estimates for other indicator pesticides are presented on Figures 6.1-12 and 6.1-13. Dieldrin exhibits the highest annual upstream loads, whereas loads for aldrin are comparatively low. This difference may reflect the fact that aldrin degrades relatively rapidly in surface water by photochemical or microbial processes (discussed further in Section 6.2). Upstream loading of these pesticides typically exhibit higher loads during high-flow conditions, with the exception of gamma-HCH where approximately 60 percent of the annual load occurs during low-flow conditions. Total surface water loads for these pesticides are dominated by the dissolved fraction (Figure 6.1-13).

Figures 6.1-14 and 6.1-15 present the upstream surface water loading rate estimates for selected metals. The highest overall loading rates are observed for zinc and copper, followed by nickel, chromium, lead and arsenic. Loading rates for mercury, which was infrequently detected (23 percent; see Table E2-5 in Appendix E), are the lowest. Loading rates during high-flow conditions for all of these metals are greater than loading rates during low-flow conditions. Further, the particulate fraction contributes more than the dissolved fraction to the total loading estimates for the majority of the metals, especially under high-flow conditions (Figure 6.1-15).

Estimated upstream total surface water loads for TBT are presented on Figure 6.1-16. There is no presentation of dissolved versus particulate fractions for TBT because the surface water data set only includes measurements of total concentrations. TBT in upstream surface water was detected only once during a low-flow sampling event. Therefore, no meaningful comparisons could be made regarding the relative loading of TBT with regard to low-flow versus high-flow conditions.

In summary, with the exception of PCBs and gamma-HCH, surface water contaminants exhibit higher upstream loading rates during high-flow conditions than during low-flow conditions. On a daily basis, loads for all of the contaminants are generally higher during high flows than during low flows. The particulate fraction represents the larger component for PCBs, PCDD/Fs, DDX, and metals. The dissolved fraction is the larger component for LPAHs, other pesticides, and hexachlorobenzene. In general, the ratios of particulate to dissolved mass loading for all surface water loading contaminants do not show large or consistent variations under different flow conditions, indicating possible conditions of equilibrium or near equilibrium, as discussed further in Section 6.2.2.

### 6.1.2 Stormwater Runoff

Stormwater loading estimates are presented in the following sections. Appendix E, Section 3.0 describes the detailed steps taken to calculate these loading estimates.

#### 6.1.2.1 Data Sources and Calculation Approach

The stormwater composite water and sediment trap data were collected in accordance with the Round 3A Stormwater FSP and Addendum (Anchor and Integral 2007b,c) and the Round 3A Stormwater Sampling Rationale (Anchor and Integral 2007d) and analyzed in accordance with the QAPP Addendum 8 (Integral 2007m).

The stormwater sampling location rationale was developed in accordance with the approach of applying representative estimates of stormwater contaminant concentrations from various land use types as described in Scheuler (1987). A land-use-based contaminant load modeling approach was used to estimate loads across the entire study area. Contaminant loading models use site characteristics (land use and percent impervious area) and land-use-specific loading rates to estimate overall loading into the receiving waters. This approach has been modified to better fit the data needs and land use characteristics of the study area, as well as the practical constraints for this sampling effort.

Samples were collected from a subset of drainage basins/outfalls within each land use category in the study area. These locations were sampled by the LWG and Port of Portland (Terminal 4) during two sampling efforts in the spring/summer of 2007 (Round 3A) and the fall/winter of 2007-2008 (Round 3B). One additional site (GE Decommissioning) was sampled by GE during the same time frame. Results from the GE investigation are also included in the overall LWG stormwater data set. In early 2008, the City of Portland collected three additional samples to supplement the residential data set, and these samples are included as well.

Loads to the study area are calculated based on composite water and sediment trap data collected from heavy industrial, light industrial, residential, parks/open space, and major transportation land use locations. Where measured contaminant concentrations fell well outside the ranges observed in the of these land use locations, the locations were defined as non-representative sites. Twenty-seven stormwater outfalls were sampled within the study area to estimate stormwater loads. In general, three to five composite water samples and one sediment trap sample were collected at each stormwater sample site. Pesticides were analyzed at a small subset of locations (8 stations) in composite water samples, but they were analyzed at nearly all locations (22 stations) in sediment trap samples.

Due to the lack of representative composite water samples for pesticides, sediment trap data were substituted for composite water results for light industrial, parks/open space, residential, and transportation land uses, as well as for 1 of 3 non-representative locations that did not have composite water data (WR-147). Additionally, composite water data were substituted for sediment trap statistics for 2 of 3 non-representative locations that did not have sediment trap data (OF-22B and WR-96).

Contaminant load estimates were generally based on approximately 100 stormwater samples across all land uses and sites. A range of 27 to 72 composite water samples

were available for heavy industrial, 10 to 16 for light industrial, 9 to 10 for residential, 2 to 3 for parks/open space, 7 to 9 for major transportation, and 3 to 5 for each non-representative location. Load estimates for pesticides were based on 26 composite water samples (from 8 stations) and 19 sediment trap samples across all land uses and sites. The number of composite water samples available for pesticide loading estimates is 12 for heavy industrial, 4 for light industrial (from one station), 3 for residential (from 1 station), zero for parks/open space, zero for major transportation, and 3 to 4 for each non-representative location. Composite sample water data were only used for loading estimates for the heavy industrial and non-representative locations. The number of sediment trap samples available for pesticide loading estimates is 11 for heavy industrial, 3 for light industrial, 2 for residential, 1 for parks/open space, and 1 for major transportation, and 1 for non-representative locations.

The stormwater analytical data set was used to generate concentration ranges for each land use and non-representative site. Both stormwater composite water samples and sediment trap chemistry data were used to provide two independent means of estimating stormwater contaminant loads. Stormwater runoff volumes draining to each model cell were then calculated for each land use and non-representative location using the City of Portland's GRID model. It was not possible to develop runoff volumes and stormwater load estimates for individual outfalls due to uncertainty of stormwater basin boundaries for many outfalls.

Loads were estimated as a product of the calculated concentration estimates and the flow rate from the 50<sup>th</sup> percentile flow year to represent a central tendency estimate of flow conditions. The annual mass loads were generated by adding the loading contributions from each land use and non-representative site for each fate and transport model segment.

#### **6.1.2.2 Uncertainty Associated with Stormwater Loading Estimates**

The primary sources of uncertainty in the stormwater loading estimates are the sample size and sampling period extrapolated to represent the composite conditions of a typical water year over the entire lower Willamette River runoff area. Specifically, data used to estimate the stormwater loads were collected during a total of 15 storm events, with each outfall sampled an average of three times. Sediment traps were left in place for 3 to 7 months during two separate sampling periods. Due to the limited time span of sampling and the known variability of stormwater, these data should be considered to represent a "snapshot" of stormwater entering the study area during the sampling period.

The methodology for calculating stormwater loading assumes that concentrations measured in individual sampled outfalls at non-representative sites are indicative of concentrations for all stormwater discharging from the site. This methodology has inherent uncertainty associated with it, as concentrations can vary significantly based on the physical characteristics of the drainage basins associated with the stormwater discharges. For example, if a drainage basin that was sampled drains a known upland



source area, the concentrations measured in this discharge may be significantly higher than stormwater discharges at the remainder of the site. Thus, this example could overestimate stormwater loading for this site. Overall, the direction of any bias in the estimates created by these uncertainties is unknown.

Other more specific factors within this particular study's methods that may contribute to the uncertainty of the stormwater loading estimates are discussed in Appendix E, Section 3.0.

### 6.1.2.3 Stormwater Loading

Table 6.1-4 presents the range of annual stormwater loads to the study area for each stormwater contaminant for both composite water and sediment trap data. These ranges are also presented on Figures 6.1-21 through 6.1-35 for each contaminant group, including ranges of the annual load estimated using both composite water and sediment trap data. Tables 6.1-5a–b present a percentage comparison of loads to the study area by land use and non-representative location for both composite water and sediment trap data.

Loads for total PCBs estimated using composite water data are slightly higher than the sediment trap estimated loads (Figure 6.1-21). The estimated loading rate for total PCBs is highest for the heavy industrial land use category as compared to other land uses, although one non-representative location contributes the greatest estimated load. A comparison of loads of individual PCB congeners is shown on Figure 6.1-22. Generally, composite water estimated loads for the various PCB components are slightly higher than the sediment trap estimated loads.

Stormwater loads for DDx pesticides are presented on Figure 6.1-24. These results indicate that the composite water estimated loads are generally within the range of loads calculated from the sediment trap data. The estimated annual loads for DDx is highest for the heavy industrial land use category as compared to other land uses; however, the highest estimated loading rates are from non-representative sites in Basin 20 (RM 6.8 to 7.4W). Loading rates from non-representative sites are based on the results from a single outfall (from the former DDT process area) that was included in summations of all land use types to represent stormwater runoff from the entire site.

Annual load estimates for total PAHs using composite water data compared well with estimates using sediment trap data (Figure 6.1-25). The estimated load for total PAHs is highest for the heavy industrial land use as compared to other land uses, with four non-representative locations contributing a substantial portion to the total stormwater load.

Stormwater loads for BEHP are presented on Figure 6.1-26. BEHP annual loads estimated using composite water data are higher than those generated using sediment trap data. BEHP estimated loading rates are highest from the heavy industrial land use areas.

Stormwater loads for hexachlorobenzene are presented on Figure 6.1-27. The results for hexachlorobenzene indicate that the sediment trap estimates are within the range of the composite water estimates. Hexachlorobenzene estimated annual loads are highest from the heavy industrial land use areas.

Stormwater loads for other organochlorine pesticides (aldrin, dieldrin, gamma-HCH, and total chlordanes) are presented on Figure 6.1-28. For each of these contaminants, the composite water annual load estimates were higher than estimates developed using sediment trap data. The estimated loads for other organochlorine pesticides are highest for the heavy industrial land use category.

Stormwater loads for metals are presented on Figure 6.1-29. Typically, the composite water load estimates for metals were slightly higher than estimates developed using sediment trap data. The highest overall estimated loads are observed for zinc, copper, and lead. Chromium, arsenic, and nickel have the next highest loads, and of the metals evaluated, mercury has the lowest. The highest estimated annual loads for metals are from the heavy industrial land use areas. Stormwater loads to the study area are presented by river mile for total PCBs, total PAHs, and DDx pesticides on Figures 6.1-30 through 6.1-35.

### 6.1.3 Permitted Point Source Discharges

Point source permitted non-stormwater discharges to the study area include NPDES-permitted discharges from commercial, industrial, private, and municipal outfalls or operations. This section presents the results of estimation of the current annual mass load of contaminants from these outfalls to the study area. The details of data compilation and loading estimation are presented in Appendix E, Section 4.0.

Both Oregon DEQ general and individual NPDES permits were considered in this evaluation. Active NPDES permits inside the study area were located using Oregon DEQ's Facility Profiler 2.0<sup>1</sup>, and the DEQ Wastewater Permits Database<sup>2</sup> was used to query the permit file numbers. There are 14 NPDES wastewater permitted discharges in the study area listed as either Individual or GEN 15A Permits. Map 6.1-1 shows the facility locations for these 14 permits. This analysis is specifically limited to permitted wastewater discharges to the study area and does not represent stormwater discharges included in stormwater loading (see Section 6.1.2) or other types of point sources.

#### 6.1.3.1 Data Sources and Calculation Approach

Permitted direct discharge loading analyses were based on water contaminant concentration data and discharge/flow data in discharge monitoring reports (DMRs), where available. These data were available for the following 10 of the 14 NPDES wastewater permitted discharges:

<sup>1</sup> Oregon DEQ's Facility Profiler 2.0: <http://deq12.deq.state.or.us/fp20/>

<sup>2</sup> DEQ Wastewater Permits Database: <http://www.deq.state.or.us/wq/sisdata/sisdata.asp>

- EOSM
- Kinder Morgan/Portland Bulk Terminal 4
- Koppers Inc.
- Starlink Logistics, Inc.
- Siltronic Corporation
- ARCO Products Company
- Kinder Morgan Liquid Terminals
- Equilon Enterprises
- Pinnacle Condominium Complex
- Univar USA.

The remaining four NPDES wastewater permitted discharges listed below were not included in the loading calculations due to insufficient data for calculations.

The facilities and the reasons they were not included are:

- Ash Grove – No flow or contaminant data reported
- Columbia River Sand and Gravel – No flow data reported and no contaminant analysis required (only TSS and turbidity monitored)
- Vigor (Cascade General) – No flow data reported on DMRs
- Hoyt Street Properties – No flow or concentration data reported.

The discharge information from these sites would be expected to increase the upper and lower end estimates of total loading to the study area for the contaminants included in their permits. However, the lack of data for these facilities is not expected to represent a significant loading data gap for any parameters.

Ranges of loading estimates were generated by considering the DMR discharge flow rates and contaminant concentration data for all the selected contaminants. Because of limited analyte lists in the DMRs and the permits, data for some parameters were not available for all facilities. Additionally, several of the selected contaminants were never monitored at any of the facilities. The results are summarized in Table 6.1-6 for the subset of selected contaminants for which data were available.

#### **6.1.3.2 Uncertainty Associated with Permitted Point-Source Discharge Estimates**

While there is uncertainty associated with the annual estimates for this loading term, the findings are expected to be reasonably representative of the relative significance of this pathway for current loading of contaminants to the study area. The primary source of

uncertainty in these estimates is the limited monitoring records available for many sites. There are four sites that could not be included in this assessment due to lack of information. If there is flow related to these permits, then discharge information from these sites would be expected to increase the upper and lower end estimates of total loading to the study area for the contaminant included in their permits.

#### **6.1.3.3 Permitted Point Source Loading**

Review of these results indicates that only a few of the analytes on the combined loading contaminant list are presented in the DMRs (for one or more permit, results are presented for DDT, select PAHs, TPH, select metals, select VOCs, and cyanide). For all of the parameters analyzed, the estimated range of results is narrow—ranging over a factor of 5. While flow volumes are relatively large for some dischargers (total permitted discharge volume is estimated to be only slightly less than stormwater runoff), the concentrations ranges are low, and the resulting loads are generally low. Because of limited volume and low contaminant concentrations, permitted point source discharges were not found to be a primary source of contaminants to the study area for those facilities and parameters for which data was available. Overall, it is expected that this loading term, as defined and assessed here, is not currently a primary source of contaminants to the study area since permitted discharges are regulated and monitored.

#### **6.1.4 Atmospheric Deposition**

Contaminant present in the atmosphere as a result of emissions from stationary, mobile, and non-point sources result in a load to the study area through the processes of dry and wet deposition. Further, persistent contaminants can travel long distances through the atmosphere from other parts of the world. Dry deposition refers to the deposition of air pollutants from atmospheric suspension in the absence of precipitation. Wet deposition refers to deposition of air pollutants from atmospheric suspension via rain or snow.

The following sections present the approach and data sources applied to generate estimates of the annual loading of selected analytes to the study area via dry and wet atmospheric deposition. Air deposition loading estimates presented here focus on dry and wet deposition directly onto the water surface of the lower Willamette River within the study area. Atmospheric deposition to land in the study area watershed which could subsequently be transported to the river via stormwater runoff is captured in the stormwater loading (Section 6.1.2).

Contaminants selected for evaluation atmospheric deposition loading are presented in Table 6.0-1. The detailed data sets, methodologies, and results for dry and wet deposition loading to the study area water surface are presented in Appendix E, Section 5.0.

##### **6.1.4.1 Data Sets and Approach**

Atmospheric deposition is the sum of both dry and wet deposition loads. Gases and particles are deposited to the ground or river surface in a process known as dry deposition, which is driven by the gravitational force on the particulate matter and the

gas aerosol. A review of studies performed to characterize dry deposition and concluded that both particulate matter and gases will contribute to the contaminant concentrations in soils and surface water bodies (USEPA 2005b). Wet deposition occurs when gases and particles are scavenged by rain droplets, freezing rain, snow, or fog droplets and are ultimately deposited to the surface.

#### 6.1.4.1.1 Dry Deposition to the River Surface

Atmospheric dry deposition to the study area was estimated based on an assumed deposition velocity, study area-specific and non-local air concentration monitoring data, and the study area surface water extent. For a given analyte, dry deposition loading (kg/yr) to the study area can be calculated as the product of the air concentration (mass/volume), the deposition velocity (length/time), and the surface area of the study area (length<sup>2</sup>). The rate of contaminant deposition to a surface (deposition velocity) is a function of atmospheric turbulence, properties of the contaminant species, and the relative reactivity of the species with the receiving surface (Seinfeld and Pandis 1998). Study area-specific or local ambient air concentration data were used, where available.

Concentration values from publicly available data sources, including DEQ and USEPA, were used for those contaminants for which local sampling data were not available. In summary, local information<sup>3</sup> was used in dry deposition calculations for all the metals, BaP, naphthalene, TPH (diesel), total PCBs, hexachlorobenzene, total PAHs, and cPAHs; exclusively external data sources were used for dry deposition estimates for the rest of the atmospheric deposition contaminants. A range of estimates was generated for the dry deposition loading fraction of the total atmospheric load. This range was based on the range of ambient air concentration results compiled.<sup>4</sup> Specific effort was made to analyze the local monitoring and modeling data for BaP and naphthalene to ensure the representativeness of the data values for dry deposition loading over the river surface (see Appendix E for details).

#### 6.1.4.1.2 Wet Deposition to the River Surface

Although wet deposition flux can be modeled, the most reliable estimation method is to collect precipitation in suitable samplers, measure the contaminant concentrations, and calculate the deposition flux corresponding to the sampling period (Reinfelder et al. 2004). Unfortunately, such data are limited; study area-specific wet deposition monitoring results were only found for total PCBs (MWH 2008) and mercury. In the MWH study, wet deposition data were collected from three monitoring stations within the study area for a 2-month sampling period spanning May through June of 2007. This study reported wet deposition loading rates calculated from the monitoring

<sup>3</sup> Local is defined here as monitoring data or modeling results for Portland, Oregon or Multnomah County, Oregon.

<sup>4</sup> The maximum value of 0.32 µg/m<sup>3</sup> BaP was determined to be an outlying value among the values from the LASAR data based on statistical analysis and was excluded from the calculation; an average value of 0.19 µg/m<sup>3</sup> was also excluded for the same reason. The following values for naphthalene were excluded from calculations based on statistical analysis: 2.16 µg/m<sup>3</sup> as one of the maximum values, 1.87 µg/m<sup>3</sup> as an average value, and 1.55 µg/m<sup>3</sup> as a minimum value. See Appendix E for more details.

concentration data (taking into consideration the field blank values). Mercury findings from Hope (2005) were considered for comparison with estimates based on the New Jersey Atmospheric Deposition Network (NJADN) data (Reinfelder et al. 2004). Briefly, the Hope study used precipitation monitoring data from Oregon Mercury Deposition Network sites (one site near Beaverton and one site near the southern end of the basin), and found wet deposition estimates comparable to those generated here by the NJADN ratio approach.<sup>5</sup> In summary, local information was used for mercury and total PCBs, and non-local/modeling data sources were used for all other contaminants.

With the exception of PCBs and mercury, for which study area-specific precipitation monitoring results were available, the monitoring results from NJADN (Reinfelder et al. 2004) were used, corrected by the ratios of 1) total atmospheric concentrations between Portland, Oregon, and Jersey City, New Jersey (where available in both), and 2) total annual precipitation between Portland and Jersey City. This approach of scaling NJADN data sets to develop wet deposition loading estimates generated only a single point estimate rather than a range because only average values were reported from the NJADN study.

#### **6.1.4.1.3 Total Deposition to the River Surface**

The total deposition loading to the study area for each selected contaminant was estimated simply by summing the dry deposition and wet deposition loading estimates. Since only central estimates could be generated for wet deposition loading, the ratio of the central estimate for wet deposition to the central estimate for dry deposition was assumed to be representative of the ratios across the range of wet deposition loading estimates. From this, upper and lower range estimates were generated for wet deposition for use in estimating the total deposition range. Where wet deposition data were inadequate to allow for estimation of even a central estimate, total loads were assigned based on the dry deposition estimates. Wet deposition estimates were unavailable for PCB TEQ, TCDD TEQ, 4,4'-DDE, 4,4'-DDT, naphthalene, total PAHs, TPH (diesel), hexachlorobenzene, aldrin, and dieldrin. The estimates are still considered to be useful based on the relatively low contribution of wet deposition to the total estimates for similar contaminants: DDx (<2 percent), BaP (~10 percent), total cPAHs (~21 percent), and total chlordanes (~16 percent).

#### **6.1.4.2 Uncertainty Associated with Atmospheric Deposition Estimates**

The lack of the study area-specific, analyte-specific, and temporally proximal data inputs for many of the contaminants places significant uncertainty on the estimates for the atmospheric deposition loading term. Specifically, local data were available only for metals, BaP, naphthalene, cPAHs (modeled), total PAHs (modeled; based on 16 individual PAHs), hexachlorobenzene, TPH (diesel), and total PCBs (modeled) for dry

<sup>5</sup>Hope (2005) calculated dry, wet, and total mercury loading rates to surface water for the entire Willamette River basin (398,000,000 m<sup>2</sup>). When scaled down to the sub-area of the basin represented by the study area (8,791,735 m<sup>2</sup>, 2 percent of the open water area estimated by Hope), Hope estimates a total atmospheric mercury load of 0.08 kg/yr. This result is slight lower than, but comparable to, the lower mercury load (0.11 kg/yr) presented here.

deposition calculations; for wet deposition calculations, local data were available only for mercury and total PCBs (limited sampling period). In the case of the atmospheric deposition loading estimates, the presented range of estimates is not expected to fully capture or represent the uncertainty associated with this term, due to significantly limited local empirical data.

The major uncertainties associated with dry deposition loading estimates are as follows:

- The limited available local atmospheric concentration data
- The simplified calculation methodology
- The uncertainty associated with selection and uniform application of a deposition velocity.

The major uncertainties associated with wet deposition loading estimates, are as follows:

- The limited local wet deposition monitoring data. Data were available only for mercury and PCBs.
- The uncertainty associated with application of precipitation correction factors to allow for use of NJADN data.

In summary, atmospheric deposition to the river surface is one of the most uncertain loading terms, primarily due to the limited availability of local atmospheric concentration and precipitation concentration monitoring data. The direction of any bias in the estimates created by these uncertainties is unknown. However, deposition to the watershed and subsequent runoff to the river is captured in the empirical stormwater runoff data set and stormwater loading estimates discussed in Section 6.1.2.

#### 6.1.4.3 Atmospheric Deposition Loading

This section presents the findings of the estimation of atmospheric deposition loading to the river surface. A qualitative discussion of atmospheric deposition to the watershed is also provided.

##### 6.1.4.3.1 Atmospheric Deposition to the River Surface

Table 6.1-7 presents the estimated ranges of annual total atmospheric deposition to the river surface for the entire study area. Figures 6.1-36 through 6.1-40 present the estimated ranges of annual loads for dry deposition, wet deposition, and total atmospheric deposition to the study area for each contaminant group.

**PCBs and TCDD TEQ** – The estimated ranges of dry, wet, and total deposition for total PCBs and TCDD TEQ are presented on Figure 6.1-36. The dry deposition fraction of the annual load represents the majority of the total annual loading estimate for total PCBs, with only less than 0.5 percent of the load attributed to wet deposition. No wet deposition data were available for TCDD TEQ.

**Pesticides** – The estimated ranges of dry, wet, and total deposition for pesticides are presented on Figure 6.1-37. The total annual loads for pesticides are dominated by the dry deposition load estimates. However, wet deposition estimates were only available for DDx and total chlordanes, and wet deposition composed 2 and 16 percent of the total, respectively. Further, DDx estimates based on NJADN estimates are lower than the 4,4'-DDE and 4,4'-DDT estimates based on ATSDR ambient concentration estimates.

**PAHs** – The estimated ranges of dry, wet, and total deposition for PAHs are presented on Figure 6.1-38. The total annual loading estimates are significantly higher for naphthalene than BaP (10 times for upper value, 3 times for central, and about the same for lower value), suggesting dominance of the LPAH fraction. Further, for all PAHs, the dry deposition fraction of the annual load represents the majority of the total annual loading estimate, with only a very small fraction attributed to wet deposition.<sup>6</sup> (Total PAH atmospheric loads are based on 16 PAHs from Oregon USEPA National Air Toxics Assessment data [USEPA 1996], which includes all of the study area PAHs except for 2-methylnaphthalene.) The PAH loading estimates are considered to be highly uncertain based on comparison with other loading term estimates. Furthermore, statistical analysis USEPA LASAR data for BaP and naphthalene indicated some data values are out of the statistical ranges that are suitable for atmospheric loading calculations, and therefore, the total PAHs values could be affected by the outliers.

**Total Petroleum Hydrocarbons (Diesel) and Hexachlorobenzene** – The estimated ranges of dry and total deposition for TPH (diesel) and hexachlorobenzene are presented on Figure 6.1-39. No wet deposition data were available for these contaminants. Furthermore, no data to support estimates of dry, wet, or total atmospheric deposition rates were available for other TPH fractions.

**Metals** – The estimated ranges of dry, wet, and total deposition for metals are presented on Figure 6.1-40. Lead, zinc, and copper exhibited the greatest total annual loading estimates by atmospheric deposition. Dry deposition loading contribution to total annual deposition was greater than the wet deposition contribution with the exception of mercury, which exhibited 7 times greater annual deposition by wet deposition. While dry deposition estimates were greater than wet deposition for the other metals, dry deposition estimates were all within a factor of 10 of the wet deposition estimates, suggesting both mechanisms are important to the overall load.

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<sup>6</sup> Wet deposition data were not available for total PAHs based on Oregon USEPA NATA data (USEPA 1996) for direct calculation of wet loading estimates; however, a closer look at the NJADN data set suggests that wet deposition is not expected to be a significant fraction of the total deposition for this chemical set. Wet deposition data were available from the NJADN study for a total based on 36 PAHs. Analysis of that New Jersey data shows that wet deposition loads are 3 orders of magnitude lower than dry deposition loads. Similarly, analysis of the 13 study area PAHs included in the New Jersey data set of 36 also shows that wet deposition loads are 3 orders of magnitude lower than dry deposition loads.



#### 6.1.4.3.2 Atmospheric Deposition to the Watershed

Contaminants that are deposited via atmospheric deposition to soils and impervious surfaces in the study area watershed may subsequently be transported to the study area via stormwater runoff. In general, for surface water bodies with relatively smaller watershed areas compared to water surface area, the total atmospheric deposition loading to the surface water is greater than the deposition loading to the watershed (Steuer 1995). But for a riverine system such as the lower Willamette River, with small surface water areas relative to the contributing watershed, atmospheric deposition to the watershed plays a greater role.

A review of available literature indicates that the relative importance of the atmospheric deposition loading term, relative to other loading terms, varies by site and by contaminant. Some studies found atmospheric deposition to the watershed to be a significant source to the surface water bodies. For instance, atmospheric deposition was found to be the dominant source term for total PCBs to the North and Baltic Seas (Struyf and Van Grieken 1993; Wania et al. 2001) and for HCH to the North Sea (Struyf and Van Grieken 1993). A recent study performed by Sun et al. (2007) in the Great Lakes region correlates average gas-phase atmospheric PCB concentrations with local population size, suggesting a strong urban source of atmospheric PCBs. Likewise, Motelay et al. (2006) found atmospheric deposition to impervious surfaces to be the most important source of PAHs to the urbanized Seine River basin near Le Havre, France. Further, one of the most recent systematic monitoring studies (the NJADN) found that direct (dry, wet, and gaseous air-water exchange) and indirect (runoff) atmospheric deposition are of major importance to the accumulation of certain elements such as mercury, and major nutrients in surface water ecosystems (Reinfelder et al. 2004). Findings from a separate, locally relevant study led by Hope (2005) of Oregon DEQ produced loading rate estimates for mercury comparable to those from the NJADN study.

Other studies found atmospheric deposition to the watershed to be less significant as a source of contaminants to surface water. A study of numerous urban U.S. streams (not including the Willamette River) evaluated the relative importance of different non-point sources of VOCs to total loading, finding that atmospheric deposition was of secondary importance for VOCs compared to the loading from urban land sources (Lopes and Bender 1998).

Contaminants deposited in the watershed surfaces are subject to a number of loss mechanisms outside of runoff transport, including leaching, degradation (biotic and abiotic), and volatilization (USEPA 2005b). Because of the complexity of the fate and transport of contaminants via stormwater runoff, a simple application of the flux rate is not appropriate for estimating loads to the study area from atmospheric deposition. Further, it is difficult to appropriately estimate the amount of deposited contaminant mass that would be transported by runoff, and even more difficult to determine how much of that entrained contaminant mass would be transported to the study area surface water given the complexity of routing and settling along the pathway. Other studies

(Deletic et al. 1997; Grottke 1987) highlight the complexity of quantitatively estimating the relative contribution of atmospheric deposition to surface water bodies. These studies note that such estimates require a detailed understanding of the geochemical process and transport fluxes specific to the urban watersheds.

The only empirical information available to assess the atmospheric contribution to the stormwater load is present in the stormwater data set. While many areas sampled as part of the LWG stormwater program have contaminant sources other than atmospheric sources, it could be assumed that samples collected from open space areas (and possibly residential areas, depending on the contaminant) represent primarily atmospheric deposition sources. Target contaminants for stormwater loading were detected in stormwater runoff in water and/or sediment trap samples in all sampled open space land-use type locations, except for 4,4'-DDD, total DDD, aldrin, dieldrin, gamma-HCH, hexachlorobenzene, naphthalene, PCB 081, PCB 126, PCB 169, and total chlordanes. PCDD/Fs and TPH were not sampled in stormwater runoff for any land-use type. Given the complexities/variables of runoff routing, adsorption of contaminants to varying surfaces, stormwater controls, a more rigorous assessment of the stormwater data set is not performed. These variables confound the utility of a direct comparison of open-space runoff to other land-use type runoff for the purposes of assessing atmospheric deposition contributions.

#### **6.1.5 Upland Groundwater Plumes**

Upland groundwater plumes flowing toward the river are a potential source of contaminants to the in-river sediments, TZW, and surface water in the study area. Seepage rate and TZW concentration data information from the nine GWPA study sites were applied to generate an estimated range of annual loads for the individual study sites. There may be additional sites that lack upland groundwater data but have complete groundwater pathways; however, such sites have not been identified or assessed.

In order to generate estimates for this loading term, observed TZW concentrations were assumed to be entirely attributable to upland groundwater as a simplifying assumption. In areas where there are both upland groundwater plume and sediment sources, contaminants detected in TZW samples may be partly or wholly attributable to contamination originating in sediment solids (partitioning into pore water). Differentiation of the origin of contaminants present in the pore water in areas with groundwater discharge and upland groundwater plumes was often not possible with the available information for certain contaminants (redox-sensitive metals, petroleum-related hydrocarbons). In such instances, the estimates of groundwater plume loading are expected to be redundant with advective loading estimates in the specific TZW study areas.

The fate and transport model addresses loading from upland groundwater plumes and from groundwater discharge through sediments somewhat differently. The model simulates the transport of contaminants within and out of the sediment bed via as

advection due to movement of groundwater, diffusion, and dispersion this transport includes partitioning. In the specific areas where there are contributions from upland plumes, an upland plume loading term is specified based on available TZW concentrations and flux estimates from filtered trident and peeper data; this additional mass is subject to the same transport processes and partitioning within the bed.

A summary of the data sets and approach used in the upland groundwater plume loading calculations, as well as a presentation and discussion of the findings is presented here. Detailed presentations of the data sets, data treatment, calculations, assumptions, and results are presented in the supporting Appendix E, Section 6.1.

#### 6.1.5.1 Data Sets and Approach

Estimates of groundwater plume contaminant loading to the study area are based on site-specific identification of potential plume discharge zones offshore of the nine TZW study sites, measured concentrations of contaminants in TZW, and measured groundwater discharge rates in potential plume discharge zones. The following data sources were used to determine these terms:

- Twenty-eight flow zone areas identified offshore of the nine TZW study sites were used to group data sets for the calculations. These flow zones are presented with discussions supporting the interpretations in Appendix C2.
- Measured shallow TZW contaminant concentrations from 150 sample locations at the nine study sites were applied to the calculations. These samples represent the complete TZW data set for the sample depth interval from 0 to 38 cm bml (see Maps 2.1-20a-l). The sampling methods used to produce this data set include small-volume peeper, Trident, and Geoprobe samplers. Both unfiltered and filtered (where available) results were included in the evaluation.
- Seventy-seven seepage meter measurements from the 28 flow zone areas were used to estimate groundwater flux for each zone. This seepage rate data is presented in Appendix C2.

As a first step, Thiessen polygons based on the TZW sampling locations were generated within each flow zone based on the TZW sampling locations to assign an area to each sample. Loading estimates were prepared for each flow zone area by summing the estimated loads for each of the sample polygons within the flow zone, using the following general equation:

$$Load_{flowzone} = \sum(C_{sample} \times A_{sample} \times UnitFluxRate)$$

Where,

$Load_{flowzone}$  = the estimated annual mass load to surface water,  $\mu\text{g}/\text{yr}$   
 $C_{sample}$  = the contaminant concentration in the TZW,  $\mu\text{g}/\text{L}$

$A_{\text{sample}}$  = the area of the Thiessen polygon associated with the given sample,  $\text{ft}^2$   
 $\text{UnitFluxRate}$  = groundwater seepage flux rate for the given flow zone,  $\text{L}/\text{ft}^2/\text{yr}$ .

A range of load estimates for each flow zone was determined by applying both the filtered and unfiltered concentrations to the calculations, as well as the average and the maximum measured seepage flux for the given flow zone. From the resulting four estimates, the highest and lowest values were assigned to designate the range. The estimate based on the average measured flux and the unfiltered concentrations was assigned as the central estimate.

The range of estimated annual loads for a given study site was determined by summing the estimated ranges for each Thiessen polygon. The ranges of load estimates for the study area were, in turn, generated by summing the estimates for each of the nine study sites.

#### 6.1.5.2 Uncertainty Associated with Groundwater Loading Estimates

The upland groundwater plume loading estimates are based on empirical, study area-specific TZW chemistry and groundwater flux data collected offshore from the nine upland sites included in the GWPA sampling program conducted as part of the RI. The range of results presented for this term is expected to be a reasonable approximation of the uncertainty in the loading estimates, though there are additional potential sources of uncertainty that may not be reflected in these ranges. Specifically, the following sources of uncertainty are acknowledged in the upland groundwater plume loading estimates:

- This assessment does not include loading from sites other than the nine study sites where empirical TZW data were collected. As described in the site selection process (Section 4.4.3.1 and Appendix C2), these nine sites represent those with a confirmed or reasonable likelihood for discharge of upland groundwater COIs to Portland Harbor. Eighty-three other upland sites reviewed during the site selection process lacked sufficient data to determine the completeness of the groundwater pathway. To the extent that a complete groundwater transport pathway to the lower Willamette River could be identified in the future at one or more of these 83 sites or other currently unidentified sites, total groundwater plume loading to the study area may be underestimated.
- The spatial resolution of the analysis is limited to the resolution of the sampling data sets, as reflected in the Thiessen polygon approach.
- There is no attempt made in these estimates to distinguish the origin of the contaminants in the TZW, and it is possible that the empirical TZW data set includes contaminants originating from sediment.

- The GWPA study design specifically targeted areas of higher seepage and higher TZW concentrations for sampling in the areas offshore of the study sites.
- The TZW concentration estimates do not account for any additional attenuation to sediments that may occur in the upper 38 cm bml.
- Sampling was conducted during the hydrologic season of highest expected groundwater flow rates to maximize the observed groundwater signal (plume concentration and flow rate). Consequently, the lower end of the groundwater signal in the discharge areas is not captured in the empirical data set.

### 6.1.5.3 Annual Groundwater Plume Loading

The estimated ranges of upland groundwater plume annual loads are presented in Table 6.1-8 at the study area scale (sum of all nine study sites) and in Table 6.1-9 for the individual study sites. Groundwater plume loads at the study area scale are also presented graphically by contaminant group on Figures 6.1-41 through 6.1-45. These group plots show the estimated loads based on filtered and unfiltered estimates to allow for comparison of these data sets. Load estimates for the individual study sites are presented graphically on Figures 6.1-46 and 6.1-47 for DDx and total PAHs only; PCBs and dioxin/furans were not sampled in TZW.

Figure 6.1-41 presents load estimates based on filtered and unfiltered TZW sampling data for DDx components. The unfiltered results are consistently higher than the filtered results for this group of hydrophobic contaminants. As discussed in Appendix D4.4, unfiltered results are likely biased high due to entrainment of sediments in the TZW samples. Figure 6.1-46 presents the DDx loading estimates for the two study sites where this contaminant was sampled in TZW.

Figure 6.1-42 presents the upland groundwater plume loading estimates for PAHs. The majority of the total PAHs load from upland groundwater plumes is from LPAHs, which is likely due their greater water solubility than HPAHs. Consistent with their hydrophobic properties, HPAHs and LPAHs show a pattern of higher unfiltered concentrations and lower filtered concentrations.

Estimates for upland groundwater plume loading of metals at the study area scale (sum of all nine study sites) are presented on Figure 6.1-43. These estimates cover a large range of values. Unfiltered/filtered loading ratios vary for different metals. The ratios for arsenic and manganese show little difference; barium, cadmium, nickel, and mercury unfiltered loading estimates are moderately greater than filtered estimates. Zinc, copper, and lead exhibit large disparities between unfiltered and filtered loading estimates. Estimated metals loads associated with groundwater discharges at individual study sites are provided in Table 6.1-9.

Figures 6.1-44 and 6.1-45 present the upland groundwater VOC and SVOC loading estimates at the study area scale (sum of all nine study sites). These plots are broken into two groups of VOCs: Group 1 includes chlorinated, non-aromatic VOCs, Group 2

contains aromatic VOCs and carbon disulfide. Among the Group 1 VOCs (Figure 6.1-44), chloroform and methylene chloride dominate the loading scale. *cis*-1,2-Dichloroethene exhibits the highest loads among trichloroethene and its daughter products.<sup>7</sup> Chloroethane exhibits the highest loads among trichloroethane and its daughter products. Among Group 2 (Figure 6.1-45), benzene loads dominate the BTEX contaminants, chlorobenzene loads are higher than those for 1,2-dichlorobenzene. Estimated VOC and SVOC annual loading are provided in Table 6.1-9.

#### 6.1.6 External Loading Summary

Table 6.1-11 provides a summary of the central estimates of external current loading to the study area for upstream surface water, stormwater runoff, non-stormwater permitted discharge, atmospheric deposition, upland groundwater plumes, and advection through subsurface sediments. The estimated annual loads for the internal transport mechanism of advection through surface sediments to surface water is also shown in Table 6.1-11 for comparison.

### 6.2 FATE AND TRANSPORT

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This section describes the physical, chemical, and biological processes that influence the fate and transport of contaminants within the study area. This discussion of fate and transport processes is organized in three main subsections, corresponding to the major environmental compartments of the study area: 1) surface mixed sediment layer and associated pore water, 2) surface water, and 3) biota. Figure 6.1-1 presents a conceptual drawing of these major environmental compartments.

#### 6.2.1 Sediment Particle and Pore Water Fate and Transport Processes

The following subsections discuss fate and transport processes relevant to select contaminants in the sediment and pore water<sup>8</sup> environment. A general discussion of organic and inorganic contaminant behavior in sediment and pore water is presented, followed by discussion of physical transport processes for these media.

##### 6.2.1.1 Contaminant Distribution between Sediment Solid and Aqueous Phases

In the sediment, the distribution of a contaminant between the solid and aqueous phases is among the most important physiochemical processes affecting its migration, bioavailability, and half-life. The equilibrium distribution of a contaminant between the dissolved aqueous phase and sorbed to sediment particles or associated organic matter is generally described by the distribution coefficient,  $K_d$ . This coefficient varies in response to environmental conditions such as pH, temperature, and salinity. Major processes and environmental factors that control this distribution are discussed below in

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<sup>7</sup> Loading estimates for trichloroethene, *cis*-1,2-dichloroethene, and vinyl chloride are dominated by results from a single TZW sample offshore of the Siltronic site. The groundwater pathway for trichloroethene is discussed in detail in Appendix C2.

<sup>8</sup> The term "pore water" is the interstitial water in the sediment within the bioactive zone.

general terms for organic and inorganic analytes. Observed partitioning ratios are compared to published literature values for relevant analyte groups for additional perspective. Finally, degradation and transformation mechanisms for contaminants in the sediment/pore water environment are also discussed.

#### 6.2.1.1.1 Organic Contaminants

Equilibrium partitioning mechanisms for organic analytes include hydrophobic sorption onto organic matter associated with the sediment, electrostatic attractions of oppositely charged ionic functional groups, and covalent bonding or complexation of ionic organic molecules with reactive surface groups. For nonionic organic contaminants (PCBs, pesticides, PCDD/Fs, PAHs, SVOCs, and VOCs), the primary partitioning mechanism is hydrophobic sorption onto organic matter. Therefore, for nonionic organic contaminants,  $K_d$  describes two-phase partitioning to the organic matter on the solid surfaces and is a function of the tendency of the contaminant to sorb to organic carbon ( $K_{oc}$ ) and the fractional organic matter content of the solids ( $f_{oc}$ ).

In addition to temperature, several factors can affect equilibrium partitioning behavior for nonionic organic contaminants:

- **Salinity** – High-salinity environments can cause increased adsorption (decreased solubility and higher observed  $K_d$  than predicted at lower salinity). This may be relevant in the highly saline sediment and pore water environment local to offshore areas on the west side of the river, between roughly RM 7 and 7.5, where pore-water salinities in excess of typical seawater have been observed. It is unlikely to be a significant factor elsewhere in the river.
- **Co-solvents** – The presence of miscible organic liquids in solution with hydrophobic contaminants can result in increased solubility (and therefore decreased  $K_d$ ) of the hydrophobic contaminant. However, this requires significant amounts of co-solvent contaminants in solution (more than 10 percent by volume [Yalkowsky et al. 1976]).
- **Colloids** – Colloids are organic and/or inorganic particles in the system defined by their behavior (tendency to remain dispersed in water, not settle rapidly, and not filter easily) and size (usually 1 nm to 1  $\mu$ m in diameter [Lyklema 1991]). Colloids represent a portion of the surface area available for sorption of organic contaminants. Because colloids can be mobile in water within a sediment matrix, they can increase the apparent concentration of the hydrophobic contaminant in the aqueous phase. Because colloids are <1  $\mu$ m in diameter, they could be present in both filtered and unfiltered water samples.
- **Characteristics of natural organic matter** – The nature of the organic matter present in the sediment can also affect the extent of partitioning, making partitioning behavior variable across different environments.

TPH is defined as the measure of all hydrocarbons that can be quantified in the carbon range from C<sub>6</sub> to C<sub>40</sub>. TPH (diesel), TPH (residual), and TPH (gasoline) are descriptive

terms for the fractions of TPH, and represent a mixture of hydrocarbon contaminants, both of natural and anthropogenic origin, with a broad range of partitioning behaviors. As such, its behavior as a contaminant group cannot be accurately characterized by a single  $K_{oc}$  value. Because the components of TPH are unknown for all sampling results, the various fractions also cannot be accurately characterized by  $K_{oc}$  values.

The partitioning behavior of TBT is strongly affected by pH and the identity of anions in solution that pair with the TBT ion (Arnold et al. 1997). Measured log  $K_{oc}$  values are on the order of 4 at pH 10 to 7, and approximately 2 at pH 7 to 3. The mean surface water pH is 7.38 (10<sup>th</sup> percentile is 6.98 and 90<sup>th</sup> percentile is 7.76). The observed pore water pH values measured in the GWPA ranged from 5.6 to 8.1.

Literature equilibrium partitioning values were compiled for the advective loading analysis presented in Appendix E. The average range in the  $K_{oc}$  values for organic analytes is 1 order of magnitude, with PCDD/Fs, TBT, and BEHP exhibiting a range of more than 2 orders of magnitude, representing substantial variability in partitioning behavior.

Site-specific empirical information to assess sediment pore water partitioning of organic contaminants is limited to the filtered TZW data set with paired surface sediment samples. This data set is limited as it focuses only on the offshore area of the nine TZW study sites, and not all COIs in sediment were analyzed in TZW samples (for example, PCBs were not analyzed in any TZW samples, and DDx and PCDD/Fs were analyzed in only a small fraction of the samples). However, because they are the only available empirical data, observed partitioning values were plotted against corresponding literature partitioning values. Observed partitioning for PAHs and DDx between filtered TZW and sediment are shown on Figures 6.2-1 and 6.2-2, respectively.

Comparisons of the literature-derived partitioning values for DDx with observed behavior are limited by the small number of sample pairs ( $n=4$ ) for which a given isomer was detected in both TZW and sediment. This limited set of observed partitioning values also spans a broad range. In contrast, the range of partitioning coefficients obtained from literature sources for individual PAHs (Figure 6.2-1) is relatively narrow, whereas the observed partitioning shows much wider ranges, especially for the LPAH constituents. The wide variability in observed partitioning may reflect multiple factors, including non-equilibrium conditions between TZW and sediment, small-scale spatial variability (sediment and TZW sample pairs were not always collocated), and/or filtered samples not reflecting truly dissolved concentrations.

#### **6.2.1.1.2 Inorganic Contaminants**

The fate and transport of inorganic species in pore water is defined by the distribution between the aqueous and solid phases. A wide range of mechanisms control the distribution of metals between these phases, most commonly precipitation/dissolution reactions and sorption/ion-exchange processes. Precipitation and dissolution are controlled by the concentration of species present both in solution and as mineral



phases. Sorption and ion exchange are controlled by a variety of factors, including electrostatic attraction, covalent bonding, and weak intermolecular attractions such as van der Waals forces.

The distribution of inorganic species between the aqueous and solid phases is controlled by a number of mechanisms that are a function of the physical, chemical, and biological characteristics of the solid-aqueous system. The characteristics most important for the aqueous solution phase include the following:

- pH
- Oxidation-reduction potential (Eh)
- Presence of competing ions
- Aqueous complexation reactions
- Ionic strength and the specific ions in solution.

The solid phase characteristics of importance include the following:

- Grain size
- Composition/mineralogy
- Sorbed organic carbon content and type
- Surface characteristics such as charge, coatings, and area.

In addition, there is a range of factors that cannot easily be assigned to one phase, such as temperature and the fugacity of gases such as oxygen and carbon dioxide.

The aqueous-solid chemistry of the sediment can be strongly influenced by microbial processes. Microbial oxidation of labile organic carbon frequently depletes dissolved oxygen in pore water, resulting in chemically reduced conditions and the production of alkalinity. Further, under anaerobic conditions, microbial processes can induce numerous environmentally relevant changes to the chemical environment, such as dissolution of iron and manganese oxide minerals and production of sulfides.

Sorption and ion-exchange mechanisms for metals can empirically be described by  $K_d$ . Unlike organic contaminants, the appropriate  $K_d$  value is not a function of  $f_{oc}$ , although organic matter can also sequester inorganic contaminants, thereby affecting the  $K_d$  value. Literature  $K_d$  values were compiled for arsenic, copper, lead, and mercury. These values were used in the calculation of pore water concentrations for estimation of loading to surface water from surface sediment via groundwater advection. This analysis and the significant uncertainty associated with the inherent assumptions are presented in Section 6.2.1.4, and the range of literature  $K_d$  values is presented in Appendix E, Table E6-6. These values show ranges of 1 to 3 orders of magnitude. The wide range in literature  $K_d$  values for metals reflects the strong, highly variable

geochemical factors described above that influence partitioning behavior in environmental systems. Considering this, literature  $K_d$  values should be considered site-specific estimates resulting from the geochemical conditions particular to individual studies. Limited site-specific empirical information, consisting of the filtered TZW data set with paired surface sediment samples for arsenic, copper, lead, and mercury, is presented for general comparison purposes on Figure 6.2-3.

#### 6.2.1.2 Degradation and Transformation Processes

A variety of abiotic degradation and transformation reactions, including hydrolysis, dehalogenation, oxidation, and reduction, can occur in aqueous systems. Hydrolysis is a reaction by which alkyl halides, esters, or ester analogs are converted to alcohols or organic acids. Dehalogenation is a reaction in which halogen atoms (such as chlorine) are removed from halogenated hydrocarbons. Oxidation and reduction are complementary reactions that involve the loss of one or more electrons (oxidation) by one chemical and the gain of one or more electrons (reduction) by another. Metals in environmental systems are subject to both oxidation and reduction reactions, depending on the particular metal, its speciation in the environment, and other geochemical conditions. Organic contaminants are subject to degradation/transformation by abiotic processes in the sediment/pore water environment, though the degradation rates are relatively slow for PCBs, BEHP, hexachlorobenzene, 1,2-dichlorobenzene, chlordanes, dieldrin, and dioxins.

Biodegradation can be a significant process for various organic contaminants found in sediments and pore water in the study area. It involves the metabolic oxidation or reduction of organic compounds and is carried out predominantly by bacteria in aqueous environments, though yeasts and fungi may also contribute to biodegradation. In general, oxidation of organic compounds occurs under aerobic conditions and reduction under anaerobic conditions, although both processes can occur under both conditions. Microbial mediated transformation of metals is only significant for mercury and lead organocompounds.

Biodegradation rates depend on chemical structure and concentration, the concentration of bacteria responsible for the biodegradation, the availability of organic matter to serve as food and energy sources for bacterial growth, and physical and chemical conditions at the site, such as temperature and oxygen level. The extent to which the organic compound is bound to particles may also affect the biodegradation rate as the bound organic compounds may be biologically less available for microbial uptake.

A wide variety of microbial species that utilize different biochemical pathways to metabolize anthropogenic contaminants have been identified. Biodegradation can proceed to full mineralization with end products of carbon dioxide and water, or an intermediate compound may be formed that is not easily further biodegraded. For example, DDT is relatively readily biodegraded to DDE, but DDE is more persistent. The susceptibility of organic compounds to biodegradation depends on several factors, such as the presence and type of functional groups, the size and

chemical structure of the organic compound, and solubility. A literature review has been completed to find appropriate biodegradation rate constants for use in the fate and transport model.

#### 6.2.1.3 Sediment Physical Transport Processes

Hydrophobic contaminants are strongly associated with sediment particles, in particular cohesive or fine-grained particles (silts and clays). As a result, the physical transport of sediments, especially silts and clays, will affect the distribution and fate of hydrophobic contaminants within the study area. Hydrophobic contaminants found in Portland Harbor include PCBs, PCDD/Fs, organochlorine pesticides, and PAHs.

Sediment movement into, within, and through the study area occurs as suspended load in the water column and as bedload along the riverbed. Cohesive or fine-grained sediments generally move as suspended load, which is defined as transport in the water column. Non-cohesive sediments (sands and coarser) typically move as bedload transport, which refers to sediment transported along or very close to the riverbed. However, a variable fraction of non-cohesive sediments moves as suspended load as a function of the flow regime, as flows increase, a larger fraction of non-cohesive sediment will move in suspension.

The movement of sediments in the lower Willamette River is controlled by both natural and anthropogenic forces that affect water movement and bottom shear stresses. As discussed in Section 3, natural flow regimes exhibit a wide range between the dry summers and rainy winters in Portland Harbor. Based on site-specific erosion velocities measured with Sedflume and modeled bottom shear stress, significant natural resuspension and movement of sediments does not generally occur at river flows below approximately 40–50,000 cfs (Section 3.1.5.2.3). Late spring through fall lower Willamette River flows are typically well below this level (see Figure 3.1-8), whereas late fall and winter flows exceed this threshold for variable lengths of time, depending on the intensity of winter precipitation events in the Willamette Basin. This strong seasonal pattern applies to the deep channel environment in the lower Willamette River which, on an areal basis, makes up much of the riverbed. Flows in nearshore, off-channel areas are severely dampened by nearshore structures, bottom drag, and shoreline configuration (such as sheltered embayments and slips). As shown on Map 3.1-11, modeled bottom shear in many off-channel areas remains relatively low even during a river flow event of 160,000 cfs. This pattern is important because most of the areas of relatively high sediment contaminant concentrations in Portland Harbor are located in off-channel areas.

In contrast to the channel environment, sediment disturbance, resuspension, and scour in nearshore areas, particularly around working piers, berths, marine terminals, and others areas with significant boat traffic, may be largely a function of anthropogenic factors, such as prop wash and boat induced waves, rather than natural factors alone. This effect may be accentuated during low flow portions of the year (late summer/fall) when river stage is low, reducing vessel drafts. This anthropogenic influence is

suggested by the time-series bathymetric measurements (Map 3.1-6), which reveal scour patterns in sheltered areas, such as Swan Island Lagoon and the International Slip, as well as very close to shore in portions of the main stem that do not experience high bottom shear forces even during high river flows.

As described in Section 3, the physical character of the lower Willamette River transitions rather abruptly near the upstream end of the study area (about RM 10) from a relatively narrow, high velocity river characterized by coarse-grained riverbed channel sediments upstream to a broader, slower river dominated by fine-grained sediments downstream. This relatively wide, fine-grained character extends to the lower end of the RI study area at RM 1.9, with the exceptions of a distinct, narrow, higher energy reach between RM 5 and 7 and a small area at the head of the Multnomah Channel; both of these areas are dominated by sands (Map 3.1-3). Measured areas of sediment scour and deposition from 2002 to 2009 (time-series bathymetry) and modeled predicted bed change during a major flood events are shown on Map 3.1-12, and depict consistent areas of erosion and deposition in the lower Willamette River during both typical (observed) and extreme (modeled) flow conditions. These areas correspond to sand-dominated (erosion) and silt-dominated (deposition) reaches. Again, this pattern applies to the deeper, in-channel portions of the river and appears to reflect the influences of natural forces. Nearshore areas are subject to a more complex mix of natural forces and smaller-scale, anthropogenic factors, such as vessel traffic, river stage variations, and in-water construction/dredging and fill activities that affect localized sediment texture and resuspension/transport patterns.

The major transport and fate processes relevant to sediment-bound contaminants are sediment transport into the study area from upstream, downstream sediment migration out of the study area (either in the main stem or Multnomah Channel), and the fate and transport of sediments within the study area, such as surface sediment mixing and resuspension, permanent burial at depth in the sediment column, and biological uptake. These processes are addressed below.

#### **6.2.1.3.1 Sediment Flux into/out of the Study Area**

Sediment enters Portland Harbor as suspended and bedload. Suspended and bedload sediment fluxes are discussed separately below.

##### **Suspended Sediments**

Suspended sediment data have been collected in the lower Willamette River across a range of hydrologic conditions. These data are described in Section 3.1.5.2.4 and show that suspended loads are strongly correlated with flow and vary from approximately 5 to 50 mg/L seasonally and annually (see Figures 3.1-26 and 3.1-25a–h). Higher suspended loads are observed on the rising limbs of the hydrographic events than on the falling limbs. Finally, a series of *in-situ* suspended particle size measurements conducted for the hydrodynamic modeling data collection effort indicate that suspended sediment particles sizes are comparable throughout the study area, with a median percentile particle diameter between 15 and 30  $\mu\text{m}$  (silt). In contrast, the median grain-

size diameter at an upriver location (RM 18) was 78  $\mu\text{m}$  (fine sand), reflecting the higher energy environment in the upper portion of the lower Willamette River (Figure 3.1-29).

The modeled HST suspended load fluxes into and out of the study area are included in Table 6.1.3. Across the modeled flow years (5<sup>th</sup>, 25<sup>th</sup>, 50<sup>th</sup>, 75<sup>th</sup>, and 95<sup>th</sup> percentiles), average total suspended sediment flux into the study area from upstream equals about 1.53 billion kg/yr, and the average total suspended sediment flux out of the study area equals about 1.26 billion kg/yr, indicating a net accumulation of about 0.28 billion kg/yr in the study area. Averaged over time, about 18 percent of the suspended material entering the harbor accumulates somewhere between RM 11.8 and 1.2. The average annual net sediment accumulation rate calculated from empirical bathymetric survey data collected between 2003 and 2009 was 0.20 billion kg/year, which is in very good agreement with the model estimates which correspond to a net accumulation of 0.19 billion kg/year. Net sediment accumulation represents a combination of new material entering the study area from upstream and some percentage of bedded sediment that is resuspended from the riverbed within the study area and then redeposited further downstream, but before exiting the study area.

#### **Bedload Sediments**

Bedload sediments move downstream along or just above the riverbed whenever near-bottom shear stresses exceed the threshold for sediment movement. Sediment entrained from the river bottom as bedload may be redeposited on the river bottom downstream, which may disperse contaminants in the sediment as they are transported downstream with the bedload. No direct measurements of bedload have been made as it is extremely difficult to measure in the field. Consequently, bedload processes are not quantified in the fate and transport evaluation.

##### **6.2.1.3.2 Sediment Fate and Transport in the Study Area**

The spatial pattern and extent of deposition and erosion in the study area was inferred from the time-series of bathymetric surveys conducted from 2002 to 2004 (described in Section 3.1.5.2.2). Based on surface and subsurface grain-size (percent fines) distribution (Maps 3.1-3 and 3.1-5) and bathymetric features (Map 3.1-9), areas of fine-grained sediment accretion appear to be dominant from RM 8 to 10, along the channel edge from RM 4 to 5, and from RM 1.5 to 3. These areas are known to be long-term sediment accumulation areas based on historical dredging needs. Upstream depressions (borrow pits) between RM 9.5 and 11, that in combination span the navigation channel, likely capture some suspended and much of the bedload sediments that are entering the system. The study area reaches between RM 5 and 7 and RM 10 and 11.8, where the river is relatively narrow, are dominated by areas of small-scale net erosion, as is the western off-channel area from RM 0 to 3 (outside bend of the lower Willamette River as it turns toward the Columbia).

Analysis of the time-series bathymetric change data presented in Section 3.1.5.2.2 indicates that during typical flow conditions only about 10 percent of the riverbed

exhibited net bathymetric changes (erosion or accretion) greater than 30 cm, but that relatively small-scale scour or accretion from about 8 cm (the limit of bathymetric resolution) to 30 cm in extent was widespread, possibly indicating that the top 30 cm of the sediment column is relatively unconsolidated and more susceptible to resuspension and erosion than deeper sediments.

The HST model prediction of bed elevation change for the high-flow flood scenario depicted on Figure 3.1-30 indicates that the spatial pattern of erosion predicted by the model for the extreme event is generally consistent with measured bathymetric change from 2002 to 2009 under more typical hydrologic conditions. However, in some areas, the magnitude of bed changes during the extreme event is dramatically greater, with erosion or deposition predicted to occur to one or more meters over observed changes (Map 3.1-12).

#### 6.2.1.3.3 Surface Sediment Dynamics

Particles that settle out or move along the bottom are subjected to a wide range of physical, biological, and chemical processes:

- **Sediment mixed-layer turbation** – Biogenic mixing by benthic infauna or bottom-foraging fish can preclude or slow consolidation of surface sediments, as can natural (such as wind waves) and anthropogenic (such as prop wash) forces. These factors can greatly complicate the spatial and temporal degree of bed erodibility. The SPI survey conducted throughout the lower Willamette River in the late fall of 2001 revealed a complex mosaic of surface sediment processes in the top 22 cm of the sediment column (the maximum depth of the SPI images) across the study area (SEA 2002b). Areas of fine-grained, low-shear sediments contrasted with coarse-grained, more compacted bottom areas. In the channel environment, these large-scale gradients in gross characteristics coincided with and helped first define the hydrodynamic reaches described in Section 3.

In some fine-grained areas, infaunal feeding pockets and worm tubes indicated that biogenic activity approached 20 cm depth. In other areas, minimal biogenic mixing activity was apparent. A well-mixed, biologically active zone appears to be on the order of 5 cm in many images, although this varied widely across the study area. Many nearshore areas showed steep onshore-offshore gradients in physical and biological conditions as a function of water depth, riverbed slope, and/or the degree of shoreline protection (embayments, structures). In some areas, layers of freshly deposited sediments exceeding 10 to 15 cm in extent were apparent. This survey was conducted during the onset of the rainy season in late November.

Overall, the SPI survey in combination with the bathymetric change data point to a dynamic surface sediment bed in much of Portland Harbor that is subjected to physical disturbance in the form of deposition or scour (on a multi-centimeter scale) due to natural and anthropogenic forces, biogenic mixing, and geochemical disturbance factors, such as methane bubble ebullition. Under

typical flow conditions, these disturbance factors appear to be limited to a maximum extent of the top 30 cm harbor-wide. Thus, frequent widespread physical and biological surface sediment mixing is likely restricted to much shallower depths over much of the study area.

- **Long-term sediment burial beneath the mixed layer** – Particles and associated contaminants that are advectively transported or buried below the mixed layer are permanently removed from the active transport system throughout most of the study area. In portions of the navigation channel upstream of RM 10.5 and between RM 5 to 7, erosion of bedded sediments to about 2 m is predicted to occur during 100-yr flood events, but this deep erosion is limited in areal extent (see Map 3.1-8b).
- **Sediment ingestion/uptake by biota** – Filter and deposit feeder organisms may actively or passively ingest particles in suspension or on the sediment bed. High densities of filter feeders can biologically enhance transfer of suspended particles to the sediment bed. Also, contaminants associated with ingested particles can enter the food web.

#### 6.2.1.4 Pore Water Physical Transport Processes

Contaminants in pore water are subject to diffusive and advective physical transport processes. These mechanisms are discussed in the following subsections.

##### 6.2.1.4.1 Diffusive Transport

Diffusion is the movement of particles or dissolved contaminants from higher to lower potential energy as represented by a difference in concentration in the case of diffusion from the pore water to the overlying water column. This is a spontaneous physical process that requires no additional energy inputs or expenditure. It is distinguished from advective transport in that it only requires a concentration gradient. Diffusive transport acts on any contaminants in solution and is therefore potentially relevant to all of the combined loading list contaminants.

##### 6.2.1.4.2 Advective Transport

Advective transport of contaminants in the sediment/pore water environment refers to the aggregate movement of contaminants by flow of pore water through the sediments to the water column in the form of groundwater discharge. It represents a transport pathway for contaminants in surface sediment/pore water to migrate to the water column, and is distinguished from the upland groundwater plume loading term described in Section 6.1.5. In certain parts of the study area, both mechanisms are likely occurring simultaneously for contaminants present in upland plumes and in sediments from other sources.

The surface and subsurface advective loading terms were assessed for the contaminants presented in Table 6.0-1. These were selected because they are likely to sorb to sediment solids and are subject to the chemical partitioning processes relevant to this loading mechanism. Loading estimates for each term were generated in units of mass

loading per year and presented for the entire study area and by river mile. Detailed presentation of data sources and approaches for advective loading is provided in Appendix E, Section 6.2.

In areas where concentrations in pore water are attributable to both upland groundwater plumes and in-river sediment sources, the plume loading and advective loading assessments may overlap, resulting in some double-counting of loads. The extent of this overlap depends on the relative magnitude of the groundwater plume concentrations versus the sediment-derived pore water concentrations based on equilibrium partitioning. The TZW plume study areas account for less than 5 percent of the study area.

#### **Study Area Annual Loading Estimates**

Study area-wide loading estimates for the subsurface and surface sediment advective loading terms are presented in Table 6.1-10. These results are also presented graphically on Figures 6.1-48 through 6.1-60, showing both surface and subsurface annual loading estimate ranges. Patterns and other observations for each of the contaminants groups are discussed in the following paragraphs.

The central estimated ranges of annual loads for total PCBs (Figure 6.1-48) are slightly higher for surface sediment to surface water than for subsurface to surface sediment. However, the difference for total PCBs (subsurface sediment advection versus surface sediment advection) is only a factor of 1.5 for the central estimates. This observation is expected due to the study area-wide higher average PCB concentrations in subsurface as compared to surface sediments. For the individual congeners, estimated load ranges were generally higher for subsurface sediment loading to surface sediments when compared to advective loading from surface sediment to surface water (Figure 6.1-49). Of the individual congeners analyzed, PCB 118 and PCB 105 exhibit the highest annual loads, whereas PCB 169 is the smallest contributor.

Advective loading estimates for PCDD/Fs (Figure 6.1-50) show a slightly greater loading from surface sediment to surface water compared to subsurface- to surface sediments. Estimates of advective loading from surface sediment to surface water are higher by a factor of 2.5 for PCDD/Fs compared to rates of subsurface partitioning to surface sediments. The OC-normalized PCDD/Fs concentrations used in the load calculations are generally similar in surface sediment and subsurface sediment, with 54 percent of the subsurface concentrations being greater than concentrations in surface sediment (Table E6-4). However, the study area-wide loading estimates are dominated by individual high surface sediment concentration values, resulting in the greater study area-wide total PCDD/Fs advective loading from surface sediment as compared to subsurface sediment.

DDD isomers comprise the largest share of the central estimate DDx advection load estimates for both surface and subsurface sediment, followed by DDT, and then DDE



(Figure 6.1-51). The study area-wide subsurface loading to surface sediment is slightly greater than the loading to surface water for each of the DDX compounds.

Advective loading estimates for other organochlorine pesticides are presented on Figure 6.1-54. Among these, gamma-HCH exhibits the highest mass loading while aldrin exhibits the lowest, possibly due to a tendency for aldrin to degrade to dieldrin in environmental systems. The subsurface to surface advective loading ratios are 0.9 for aldrin, 1.15 for dieldrin, 0.6 for gamma-HCH, and 2.5 for total chlordanes.

Total PAHs annual load from both the surface and subsurface sediments is dominated by LPAHs (Figure 6.1-52). HPAHs exhibit slightly higher surface sediment loading to surface water relative to subsurface loading to surface sediments. In contrast, naphthalene and total PAHs loading from subsurface to surface sediment is greater than to surface water. The estimated PAH loading from subsurface to surface sediment is greater than loading to surface water by a factor of 3 for the central estimate.

The range of advective load estimates for BEHP are presented on Figure 6.1-53. The 5 orders of magnitude range in the estimated loads is a direct reflection of the large range in the literature  $K_{ow}$  (octanol-water partitioning coefficient) values. These estimates show slightly more surface sediment loading (by a factor of approximately 2 for the central estimates).

Advective loading rate estimates for arsenic, copper, lead, and mercury are presented on Figure 6.1-55. Arsenic and copper show the highest study area-wide loading, followed by lead. Mercury exhibits the lowest loading estimates, with central estimates 4 orders of magnitude less than the corresponding estimates for arsenic and copper. Copper, mercury, and lead surface and subsurface loadings were essentially equal (ratios of 0.8 to 1.1, respectively), and representative of the fairly even distribution of surface and subsurface sediment concentrations over approximately 60 to 70 percent of the study area (Table E6-4). The arsenic surface sediment to surface water loading estimate is 2.2 times greater than the subsurface loading to surface sediment estimated loading.

Estimated TBT advective loads (Figure 6.1-56) vary over 3 orders of magnitude from the lower to upper estimates. The subsurface-to-surface sediment loading estimate is 3.7 times higher than the surface sediment loading to surface water for the central estimates.

In summary, study area-wide advective annual loads from subsurface sediment to surface sediment were higher than advective loading from surface sediment to surface water for PCBs, DDX, LPAHs (and total PAHs, which are dominated by LPAHs), BEHP, arsenic, total chlordanes, and TBT. The opposite was true for PCDD/Fs, gamma-HCH, and HPAHs. There was little difference between the surface and subsurface advective loading estimates for aldrin, dieldrin, copper, mercury, and lead. These differences are a direct reflection of the patterns of relative OC-normalized concentration of each contaminant in surface as compared to subsurface sediment.

### **Annual Loading by River Mile**

Figures 6.1-57 through 6.1-60 present annual surface sediment and subsurface sediment advective loading for each river mile in the study area for total PCBs, total PCDD/Fs, DDX, and total PAHs. Given the nature of the analysis, variations in river-mile-scale annual load estimates are indicative of variations in sediment contaminant concentrations and organic carbon content. Groundwater flux rates and assumptions of equilibrium behavior were held constant over the entire study area in advective loading calculations.

The sediment advective loading pattern for total PCBs is fairly complex. As shown on Figure 6.1-57, the highest central estimates of advective loading from subsurface sediment to surface sediment are observed at RM 8 to 9. The highest annual surface sediment loading estimates to surface water are observed at RM 9 to 9.9.

Subsurface sediment advective annual loading to surface sediment, and surface sediment advection to surface water for total PCDD/Fs (Figure 6.1-58) are fairly consistent across the study area, with the exception of higher annual loading estimates to surface water from RM 7 to 7.9.

The highest surface and subsurface sediment advective loads for DDX are predicted at RM 7 to 7.9 (Figure 6.1-59). Subsurface-to-surface sediment advective loading annual estimates are comparable or greater than the estimates of surface sediment advection to surface water in all river miles except RM 9 to 10 and RM 11 to 12, where the load from advection to surface water is somewhat greater.

The highest subsurface advective loading to surface sediment for PAHs (Figure 6.1-60) is observed at RM 6 to 6.9. The maximum load estimates for surface sediment advection to surface water are observed at RM 5 to 5.9. Subsurface-to-surface sediment advective loading estimates are comparable to or greater than the load estimates for surface sediment advection to surface water in all river miles except RM 5 to 6 and RM 9 to 10, where the load from advection to surface water is somewhat greater.

#### **6.2.1.4.3 Uncertainty Associated with Advective Transport Estimates**

There is uncertainty associated with the advective annual load estimates related to applied assumptions (including equilibrium behavior of all contaminants and uniform groundwater discharge rates), as well as the data sets used in the calculations (literature equilibrium partitioning coefficients, and roughly estimated groundwater discharge rates).

The primary uncertainty related to equilibrium is the assumption of equilibrium in all parts of the complex sediment/pore water environment. This calculation fails to capture reaction kinetics and the sorption-desorption-resorption dynamics that occur in advective transport through sediment. For example, to the extent that non-equilibrium conditions may exist in the pore water environment as a result of kinetic limitations on desorption from contaminated sediments, the assumption of equilibrium will overstate

pore water concentrations and advective loading rates. Beyond the assumption of equilibrium, the study area organic carbon associated with sediments may differ in character from that defined by the range of literature  $K_{oc}$  values. Likewise, the location-specific chemical and geochemical conditions (redox, pH, ionic strength and composition, sediment matrix composition, etc.) likely differ in character from those associated with the applied specific literature values. Further, this assessment ignores any chemical or biological transformation processes that may occur in the migration process.

There are a number of significant uncertainties related to the groundwater flux rate estimates. First, they are based on the limited available upland data and not on groundwater modeling of the area or direct measurement of seepage rates representative of the entire study area. Second, the groundwater advection rate estimates rely on a simple and conservatively high cross-sectional area. Third, the advection rate estimates apply a projection of the sediment surface area to represent the actual sediment surface area (thereby increasing the unit discharge estimate). Finally, the assumption of a uniform groundwater discharge rate for the entire study area does not capture the spatial variability that likely exists throughout the study area. The discharge rates are assumed to be constant in time and do not account for variability caused by seasonal recharge patterns, changes in river flow rates and stages, and tidal fluctuations (tidal pumping).

Among all seepage meter locations where net positive average advective groundwater fluxes were measured, the largest net negative recharge rate during a rising tide was offshore of the Siltronic site (Appendix C2). At this location, the negative recharge period covered roughly 9.5 hours, with an average seepage rate of  $-6.7$  cm/day. This corresponds to a net negative seepage flux of 2.65 cm into the sediment bed over the 9.5-hour tidal recharge period. Assuming sediment porosity of 25 percent, the maximum depth of influence for this period of negative seepage would be approximately 10.6 cm before the direction reversed to positive discharge with the tidal change. Although tidal pumping may in some instances lead to increased loading of contaminants from the sediment bed to the water column by introducing relatively clean surface water into the uppermost several centimeters of the sediment bed with each tidal cycle, uncertainty in the loading estimates due to this effect is expected to be minor compared to the other sources of uncertainty inherent in these calculations.

The large range in most of the estimates presented in the following subsection reflects the range in literature equilibrium partitioning coefficients. These large-scale estimates of advective annual loads are considered to be uncertain, but useful for general comparison to other loading terms for each contaminant.

### 6.2.2 Surface Water Fate and Transport Processes

Fate and transport processes for contaminants present in the dissolved phase and sorbed to suspended solids include partitioning between surface water, air, and suspended sediment, physical transport of surface water and suspended solids, and physiochemical and biological processes are described below.

#### 6.2.2.1 Contaminant Distribution between Surface Water and Suspended Sediment

The observed partitioning between surface water (filtered) and suspended sediment for surface water samples for PAHs, PCDD/Fs, PCB homologs, DDX, and other pesticides is presented on Figures 6.2-4 through 6.2-8. Limited site-specific information for arsenic, copper, lead, and mercury is presented for general comparison purposes on Figure 6.2-9. These figures also show the literature  $K_{oc}$  and  $K_d$  values compiled for use in the advective loading assessment. For most contaminants, the observed partitioning between suspended sediment and filtered surface water spans a wider range than the literature  $K_{oc}$  values. The source of this variability is unknown, but may be attributable to non-equilibrium conditions between surface water and suspended sediment, errors introduced by the estimation method for the  $f_{oc}$  content of suspended sediment (see Appendix E, Section 2.4), filtered surface water samples not reflecting truly dissolved concentrations, or a combination of these factors.

Based on visual inspection of the information presented on Figures 6.2-4 through 6.2-9, the central part of the range of observed partitioning values corresponds with the CT in the literature values for PCB homologs, PCDD/Fs, and pesticides (excluding DDX). In contrast, the observed partitioning in the data set generally appears to be biased high relative to literature  $K_{oc}/K_d$  for PAHs and, to a lesser degree, DDX pesticides and metals. McGroddy et al. (1995, 1996) noted that only a small fraction of PAHs present in bulk sediment from Boston Harbor appeared available for equilibrium partitioning; empirically derived log  $K_{oc}$  values were significantly greater than literature values, specifically for phenanthrene and pyrene. They concluded that PAH compounds associated with soot particles typical in many coastal and estuarine areas may be less available to exchange with the pore water than suggested by the literature, and that equilibrium partitioning models overestimated the pore water and desorption aqueous-phase PAH concentrations by as much as a factor of 100. Thus, modeled PAHs concentrations may be overestimated when based on literature values for  $K_{oc}$  and an assumption of equilibrium partitioning.

#### 6.2.2.2 Physical Transport of Contaminants in Surface Water

Advection is the flow of river water in response to gravitational forces, and is the primary mechanism for transport of surface water and its load of dissolved and particle-bound contaminants. River flow is quantified using water velocity and discharge. Water velocity is dependent on the slope, shape, and physical characteristics of the riverbed and has the dimensional units of length/time (ft/s). Discharge represents the quantity of water passing a specific location within a specific time interval. It is calculated as the average velocity times the cross-sectional area of the river, and has the dimensional units of volume/time (cfs or L/yr). The surface water mass flux of a contaminant is the product of the concentration and the volumetric flow rate of the river, producing dimensional units of mass/time (kg/yr), as calculated in Section 6.1.1.1 for surface water load estimates.

The dominant direction of water flow in the lower Willamette River is downstream along the hydraulic gradient. However, the flow direction reverses on flood tides during low-flow periods (see Section 3.1.4.3). Upstream flow has been identified as far upstream as RM 12.8 during low-flow conditions (Figures 3.1-22a–h; Caldwell and Doyle 1995).

Lateral and vertical movement of contaminants in surface water occurs primarily as a result of turbulent dispersion, and to a lesser extent as a result of mixing and diffusion resulting from concentration, thermal, and density gradients. The velocity of river water is greatest near the center of the river and decreases toward the sides and bottom. These differences in velocity result in velocity shear, which gives rise to eddies. These may also be caused by channel irregularities, including structures in the water. These processes serve to mix the water and dilute contaminant concentrations as they move away from the source. The suspended load of particle-sorbed contaminants can also decrease due to settling of particles to the riverbed sediment surface.

Sources of contaminants to surface water, such as industrial point discharges or groundwater plume discharge areas, can result in plume formation as the contaminants mix with and diffuse into river water flowing downstream. Mixing patterns and plume sizes depend on differences in density between the effluent and river water, the depth, velocity, and turbulence of the river, and any density stratification of the river itself. Density is a function of the temperature and salinity of the water.

Suspended particles provide an important vehicle for exchange of contaminants between the sediment bed and surface water. Suspended particles can be derived from mineral sources, including eroded and weathered rock, or from organic sources, such as decaying plant material or plankton. The density of mineral particles is generally 2 to 3 g/cm<sup>3</sup>, whereas the density of organic particles is close to the density of water (1 g/cm<sup>3</sup>). The entrainment and settling of suspended particles are functions of river flow rate, particle size, particle shape, and particle density. The sediment-carrying capacity of river water increases with increasing stream flow and turbulence, which vary spatially as well as temporally. Stream flow, turbulence, and TSS loads are greater in areas where the river is narrower, and throughout the river during high-flow events. Within the water column, suspended particle concentrations generally decrease from the riverbed to the water surface. TSS in surface water across the study area increases with increasing flow rate. The range of TSS as a function of flow rates decreases by RM 2, where river turbulence decreases.

#### **6.2.2.3 Physiochemical and Biological Attenuation Processes in Surface Water**

In addition to equilibrium partitioning, several physical, chemical, and biological processes can result in transfer of contaminants found in surface water between abiotic media, or in degradation/transformation reactions. These include chemical precipitation, volatilization, abiotic degradation (chemical reaction or photolysis), and biodegradation. With the exception of volatilization and photolysis, these processes

also generally pertain to sediment aqueous and particle interactions previously discussed in Section 6.2.1.2.

Volatilization is the transfer of contaminants dissolved in surface water to the atmosphere, and is most important for small organic molecules such as VOCs. It is dependent on water and air temperature, dissolved concentration, and vapor pressure. Water turbulence and wind velocity at the air/water interface will also affect volatilization rates. Volatilization typically decreases with increasing molecular weight. Additionally, various forms of mercury and organolead compounds may also volatilize from the water column. Equilibrium partitioning between dissolved volatilized phases is defined by the Henry's law constant (H).

Photolysis degradation or transformation reactions occur in response to absorption of solar energy, and can occur either directly or indirectly. Direct photolysis is the breaking of molecular bonds by electromagnetic radiation, particularly high-energy ultraviolet radiation. Indirect photolysis involves formation of a reactive species such as a hydroxyl radical or oxygen singlet, which subsequently reacts with an organic molecule. Examples of indirect photolysis include cleavage of aromatic rings, hydrolysis, hydroxylation, or dechlorination reactions. The degree to which photolysis occurs is affected by the depth and turbidity of the water, and by the intensity and angle of incidence of light. It can be significant for aldrin, PAHs (especially LPAHs), PCP, TBT, and organolead compounds. Additionally, contaminants sorbed to labile organic carbon can be released to the water column through degradation of the dissolved/suspended organic matter. PCBs and PCDD/Fs are also subject to photolysis in surface water, though the process is considered to be minor for PCBs and is only relevant to PCDD/Fs near the water surface (USEPA 1994).

### 6.2.3 Biota-Related Fate and Transport Processes

A number of processes govern how organisms living in the lower Willamette River are exposed to contaminants and how contaminants are transformed, excreted, or stored in tissue. Organisms living in the lower Willamette River take up contaminants through physical, chemically- and biologically-mediated processes, including transfer of waterborne contaminants across gill structures or other tissues, consumption of prey, or ingestion of sediment. Organisms can modify the contaminant burden in their tissues through growth, reproduction, excretion, metabolic transformation, or sequestration. Some contaminants are transferred among organisms through trophic interactions, resulting in increases in concentrations of some contaminants at higher trophic levels.

PCBs, pesticides, PCDD/Fs, and PAHs, and similar hydrophobic contaminants, are likely to be associated with organic materials (lipids in tissues, dissolved or particulate carbon in the surface water, pore water, and sediment). However, some metals (lead and zinc) also tend to associate with organic and inorganic solids because the geochemical properties, such as ionic charge, governing their behavior tend to promote sorption.

Once released to the aquatic environment, contaminants enter the food web in a number of ways; the process is not sequential in that all trophic levels can interact with abiotic media. The behavior of contaminants within an aquatic food web is briefly described below.

Primary producers such as phytoplankton and plants take up contaminants primarily through diffusion from water. The lipid content of phytoplankton also serves as a substrate for the partitioning of organic compounds. Metabolic byproducts of phytoplankton contribute to the colloidal material in the water column, which can also serve as a binding substrate for dissolved contaminants. These colloidal materials can be directly utilized by bacteria, other phytoplankton, and zooplankton, serving as an additional uptake and transfer mechanism for recycling contaminants within the water column food chain. Zooplankton prey upon phytoplankton and other zooplankton, further recycling contaminants within the water column. More complex aquatic organisms (invertebrates and fish) can take up dissolved- or colloidal-bound contaminants from surface water and pore water across gill membranes, skin, and other permeable tissues, such as the mantle in clams (shells, exoskeletons, and scales are less permeable). Sediment surfaces may be coated with bacteria and bacterial slimes, natural organic polymers, and other amorphous organic molecules that serve as binding sites. Finer-grained sediments have a greater surface area-to-volume ratio and thus have a greater organic carbon content and contaminant concentrations.

Once sediment or prey is ingested by invertebrates and fish, the rate of contaminant absorption across gut membranes is affected by the size of the molecule (larger molecules are more difficult to transfer across membranes), concentration gradients between gut content and surrounding tissues, acidity of the gut, and other physical/chemical conditions in the gut. Absorbed contaminants may undergo various metabolic processes that change the chemical structure and properties.

Once absorbed, metals that are not excreted may be stored in calcium carbonate matrices (invertebrates) or bone (vertebrates), which tend to reduce the reactivity of the metal. Organic contaminants that are not metabolized tend to be stored in organs or fatty tissues, including gametes. These stores can be released within the aquatic and terrestrial food webs when these organisms are ingested by others, upon their death and decomposition, or by transfer to their offspring.

### **6.3 UPPER STUDY AREA SEDIMENT CORES**

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This section details contaminant concentration with depth from cores collected in known depositional areas at the upper end of the study area. Three cores were collected in Round 3A in three different known depositional areas based on the time-series bathymetric data (Figure 6.3-1). This sampling effort is detailed in the Round 3A FSP (Integral 2006q), and the full data sets are presented in the corresponding data report (Integral 2007g,h).

The objectives of this sampling effort were to analyze both radioisotopes and conventional/contaminant chemistry at uniform and continuous depth intervals in long-term depositional areas expected to act as natural sediment traps. Because of the location of these cores, these data allow inferences to be made about deposition rates and the chemical composition of sediments settling out in the upper study area. Two of the three stations sampled, RC02-2 at RM 10.9 and RC01-2 at RM 10.5, are situated in formerly excavated borrow pits with mudline depths well below the authorized channel depth of -40 ft CRD. The third station, RC483-2 at RM 9.6, is located in the main channel on the large shoal that occurs along the western half of the channel in this area.

Detailed evaluation of the radioisotope data from these cores is provided in Anchor (2007e). Because of the heterogeneous origins of the sediments making up the deposits, the radiochemical data did not support the assignment of a timeline to the sediment profiles. However, empirical data on the history of the borrow pits as well as the shorter-term LWG time-series bathymetric data support overall sedimentation rates of approximately 1.5 ft/yr (45 cm/yr) at RC02-2 and 1 ft/yr (30 cm/yr) at RC01-2 (Anchor 2007e). These rates represent a long-term average over multiple years. The actual sedimentation in any given year is likely variable and may be higher or lower than this net long-term average.

The remainder of this section focuses on the conventional and contaminant chemical data measured in these cores with depth. These core samples were sectioned and sampled in 30-cm segments from the mudline to the bottom of each core. This allows inferences to be made about the quality of material entering and settling in the upper portion of the study area over time.

### 6.3.1 Upper Study Area Depositional Core Sediment Quality

The locations of the three depositional cores in the upper study area are presented on Figure 6.3-1. As noted above, RC483 is a shoal area on the western side of the channel at RM 9.6. RC01 and RC02 are located in dredged borrow pits on the western side of the channel at RM 10.5 and RM 10.9, respectively. Summary statistics for all core segments for all three cores combined are provided in Table 6.3-1 and for each core individually in Tables 6.3-2 through 6.3-4. A range of contaminants plus grain size and percent TOC are included in these tables. The upper study area depositional core data evaluation that follows focuses on the physical nature of the cores and the measured concentrations of four contaminants, total PCBs (Aroclors)<sup>9</sup>, TCDD TEQ, DDx, and total PAHs.

#### 6.3.1.1.1 Physical Texture

Figure 6.3-2 shows the core log physical description for each core. Core recovery ranged about 260 cm at RC01 to 330 cm at RC02. All three cores show a general pattern of an upper silt layer deposited over a distinct sand interval, which is

<sup>9</sup> PCB congeners were not analyzed in these core samples, so total PCBs concentrations are based on Aroclor data only.



approximately 40 cm thick at RC01 and approximately 15 cm at RC483 and RC02. This subsurface sand layer may represent coarse-grained material deposited during the most significant, recent high-flow event on the lower Willamette River (approaching 200,000 cfs; see Figure 3.1-8) that occurred in December 2005/January 2006. The overlying 30 to 40 cm of silt would be consistent with an approximate 1-year time frame from cores collected in February 2007. Both the estimated long-term sedimentation rate of 30 to 45 cm/yr based on the borrow pit in-filling data noted above and the measured 2002 bathymetric change at stations RC01 and RC02 which averaged 38 cm/yr and 34 cm/yr, respectively, over the 7-year period from 2002 to 2009. Below this sand layer in each core, there is a thick silt layer that varies somewhat in character between the three cores. The silt layer is interbedded with fine sand lenses in RC483, the shoal location at RM 9.6, and RC02, the borrow pit at RM 10.9. In RC02 the texture becomes increasingly sandy below 240 cm down to another distinct sand layer at 315 cm. It is very possible that this deep sand layer reflects the high-flow event (approaching 250,000 cfs; see Figure 3.1-8) that occurred in the lower Willamette River during the winter of 1998/1999. The 315 cm of accumulation over the 8-year period from this horizon to 2007 equals an average sedimentation rate of 39 cm/yr. This is consistent with the long-term sedimentation rates estimated for this area. At RC01, the subsurface silt layer exhibits thick organic beds below 90 cm, suggesting some heterogeneity in the quality of material settling out within this portion of the river.

Figure 6.3-3 shows the 30-cm composite interval results for grain size and TOC with depth for each core. Grain size with depth is consistent with visual core log information at RC483 and RC02 with fine-grained sediments (60 to 80 percent fines) dominant throughout the core except for where distinct sand layers are evident. RC01 is more variable in texture with depth but does show the distinct shallow subsurface sand lenses. Consistent with the organic debris observed at depth in RC01, TOC values are somewhat higher in this core below 90 cm (exceeding 3 percent in most intervals) than in the other cores.

#### 6.3.1.1.2 Contaminant Vertical Profiles

Figures 6.3-4 through 6.3-7 present vertical profiles of the bulk sediment chemistry concentrations on both a dry-weight and TOC-normalized basis for total PCBs, TCDD-TEQ, DDx, and total PAHs in each core. Non-detects are plotted at the full detection limit with an open symbol. Selected summary statistics (using detected values only) for the data from all three depositional cores combined, as well as each individual core, are provided in Table 6.3-5.

The vertical profile data across the four analytes show some general trends. Dry-weight contaminant concentrations in all three cores vary with sediment grain size and TOC, with lower concentrations for all contaminants measured in the sand layers. The measured values for all analytes is generally low measured values for all of these analytes across all cores, with corresponding minimal vertical gradients within and between cores. Some exceptions to these general trends include a dioxin and PCB spike in the 180-to-210 cm interval in RC01, which may correlate with organic-rich beds in

the silt layer. Assuming average sedimentation rates, this horizon may correspond to the atypically low-flow water year in 2001 (Figure 3.1-8). Another exception is the notably higher total PCBs levels at RC02 (RM 10.9) compared with RC01 (RM 10.5) and RC483 (RM 9.6). While the levels in RC02 average less than 20 µg/kg, this compares with mostly undetected values in the cores farther downstream. This difference appears to reflect the influence of the proximal source or sources of PCBs on the east side of the river at RM 11.5 (see Section 5.2). Finally, slight vertical trends with concentrations increasing with depth are evident in the TOC-normalized PCB and possibly the TOC-normalized PAH data at RC02.

The vertical profiles of the four indicator contaminants measured in three cores from known depositional areas in the upper portion of the study area show relatively low concentrations for all contaminants and minimal gradients with depth within each core and between cores. The farthest upriver core at RM 10.9 exhibits slightly elevated PCB concentrations compared with the other cores, and this may reflect the influence of the elevated PCB concentrations detected just upstream at RM 11.5E. Otherwise, the relatively low contaminant concentrations measured in these known depositional area cores appear to reflect the quality of sediments entering and settling out in the upstream portion of the study area over approximately the last 10 years. This trend may be representative of the periods of significant sediment deposition and accumulation in the lower Willamette River associated with conditions (higher flows, precipitation) that bring large volumes of sediment to the river, and this would act to dilute the relatively localized sources of contamination in bulk sediment deposits.

### 6.3.2 Upper Study Area Deposition Rates

Based on the harbor-wide measured riverbed elevation changes over the 7-year period from 2002 to 2009 (see Map 3.1-6), the net sediment accumulation rates in these upstream borrow pits at RM 10.9 and 10.5 are estimated to be approximately 41 and 31 cm/yr at RM 10.9 and 10.5, respectively. This is consistent with the 19 year estimates noted previously in Section 6.2. The borrow pits themselves, spanning the channel in this reach and presenting a relatively larger channel cross-sectional area (because of their greater depth) than in reaches immediately upstream, provide conditions that promote sedimentation. However, based on the bathymetric survey data, sedimentation rates in this portion of the lower Willamette River outside the borrow pits, such as the large shoal that occupies the western portion of the navigation channel from RM 8 to 10, are comparable in scale (31 cm/yr at the maximum shoaling point at RM 9.6, see Map 3.1-10). This shoal area has historically required regular maintenance dredging (see Section 3.2.3.1.13).

The long-term sedimentation rate observations noted above apply to the study area channel environment. Based on bathymetric change, SPI data, and limited radioisotope sampling for MNR assessment (Anchor 2005b), nearshore and off-channel areas do not appear to accumulate sediment at these rates. Short-term active sediment deposition and resuspension are indicated by these data sets, likely due in many areas to anthropogenic activity. Seasonal (rainy season) inputs of fine-grained sediments in

areas adjacent to the channel are also evident. However, seasonal comparison of surface sediment textures at similar locations in the spring versus the fall suggests that some nearshore deposits can be remobilized over time and dispersed (WEST and Tetra Tech 2009), minimizing net accumulation rates. These observations are supported by the radioisotope data from four nearshore areas in 2004 (Anchor 2005b), which show well-mixed surface sediment layers and calculated net sedimentation rates of approximately 1 cm/yr.